Wild Turkeys in the Urban Matrix: How an Introduced Species Survives–and Thrives–in a Multifunctional Landscape

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Abstract

An understanding of how species responds to urbanization is important for conservation and management of possible human-wildlife conflicts. Wild turkeys (*Meleagris gallopavo*) have recently successfully entered many urban landscapes, however their apparent success remain poorly understood. Most studies of wild turkeys have occurred in forested or agricultural landscapes. I estimated several important demographic, home range, and habitat use behaviors for wild turkey in areas of varying degrees of urbanization in the Minneapolis-St. Paul, Minnesota, USA, metropolitan area. My research objectives centered on providing the first information on urban wild turkey ecology, including: 1) assessing urban wild turkey nesting behavior and possible changes to reproductive measures, 2) investigation of urban wild turkey survival and the influence of local mortality agents, and 3) assessing urban wild turkey home range characteristics and habitat use.

I captured and equipped 60 female wild turkeys with back-pack style VHF radio transmitters during 2010-2013. Monitored female wild turkey reproductive measures and nest survival were remarkably similar both among my study areas and previous rural wild turkey research. For all monitored females across all study areas and years, first nesting rate was 73.7% (n = 57), average date of onset of incubation was 2 May (n = 42), and hatch rate was 84% (n = 26). For all monitored females across all study areas and years mean clutch size was 10.2 (n = 42), and differed by study area (χ^2 = 8.30, DF = 2, P = 0.02). For all monitored females across all study areas and years nest survival rate was

0.56 (n = 42). Monitored nests tended to have high visual concealment at the nest bowl, with a strong trend for habitat variables related to vegetative density and height at the nest bowl scale, and distance to open water on the greater landscape.

Across all study areas and all female wild turkeys in 2010-2013 the annual survival rate was 0.43 (CI = 0.32 – 0.58; n = 55). Across all study areas and all female wild turkeys in 2010-2013, seasonal survival rates were as follows: 1) spring survival rate was 0.61 (CI = 0.50 – 0.75; n = 55); 2) summer survival rate was 0.83 (CI = 0.71 – 0.96; n = 34); 3) autumn survival rate was 0.89 (CI = 0.75 – 0.99; n = 28); and 4) winter survival rate of 0.96 (CI = 0.89 - 1.0; n = 25). During the brooding seasons of 2010 through 2012, an estimated 216 poults successfully hatched. Combined poult survival rate to 2 weeks post-hatch was 0.35, declining to 0.26 4-weeks post-hatch. Overall, mammalian and avian predation accounted for 63.3% of all observed female mortalities, followed by vehicle strikes (23.3%), harvest (3.3%), and unknown causes (10.0%). Predation remained the leading cause of mortality regardless of age-class, although predation tended to be higher in female adults (61.5%) than juveniles (47.1%).

Across all study areas and all female wild turkeys, average annual home range size was 41.3 ha (n = 28). Annual home range size for suburban females (64.5 ha, n = 9) was larger than rural (38.0 ha, n = 11) or urban females (19.6 ha, n = 8), with home range size differing between study areas (χ^2 = 12.26, DF = 2, P = 0.002). Spring/summer home ranges included both females that attempted to nest, brooding hens, and non-

reproductively active females. Across all study areas and all female wild turkeys, average spring/summer home range size was 26.4 ha (n =37). Spring/summer home range size for suburban (44.8 ha, n = 11), rural (23.0 ha, n = 17), and urban females (10.3 ha, n = 9) did not differ. Across all study areas and all female wild turkeys average autumn/winter home range size was 25.1 ha (n = 28). Autumn/winter home range size for suburban (30.9 ha, n = 9), rural (28.6 ha, n = 11), and urban females (13.8 ha, n = 8) did not differ.

Habitat use by wild turkey populations in urban settings relied heavily on ‘natural-like’ habitat, as well as on developed, human-dominated areas. For this study ‘natural-like’ habitat (i.e., parkland, conifer tree) was predictive of spring/summer habitat use and developed habitat (i.e., residential areas, agricultural) was predictive of autumn/winter habitat use. These range shifts are likely linked to resource availability and specific habitat availability (i.e., nesting and brood habitats).

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Chapter 1: Introduction

Over the past 50 years, human population growth and expansion have proceeded at an unprecedented rate, impacting up to half of the planet's surface with some form of human activity (Vitousek et al. 1997). In the continental United States, for instance, urban and suburban land use increased from less than 1% and 5% respectively in 1950 to 2% and 25% by the 1990s (Brown et al. 2005). A major element of human land use change is the large-degree of habitat alteration associated with intensified human uses (Turner et al. 1995). As a consequence of these modifications, urbanization is a leading cause associated with threats to native species (Czech et al. 2000).

Of all forms of land use change governed by human actions, none alter natural landscapes or influence wildlife to a greater degree than urbanization (McIntrye and Hobbs 1999, Czech et al. 2000, Marzluff and Ewing 2001). The process of urbanization transforms 'natural' landscapes, such as forest, prairies, and wetlands, to human-dominated areas of residential, commercial, and industrial use (Grimm et al. 2000, McKinney 2008, Garden et al. 2010). Conversion to urban land use increases threats from habitat loss, fragmentation, and degradation, which can impact species by reducing habitat area, changing landscape structure, and increasing physiological stresses (McDonnell and Pickett 1990, Savard et al. 2000, Adams et al. 2005). For avian species, urbanization influences individuals and populations through shifts in predator communities, increased human disturbance, and reproductive measures (Chace and Walsh 2004). However, in the

long-term, how a species responds to differing landscapes depends on their life-cycle requirements and sensitivity to the specific land-use changes.

Traditionally, wildlife conservation and management addressed the issue of land-use change by focusing heavily on conservation of biodiversity in wilderness areas, parks, and other natural areas (Marzluff and Rodewald 2008). Recently many public agencies have increased their interest in restoring and managing habitat within the urban landscape. Similar to rural areas, urban wildlife provides economic and recreational benefits, and an increased quality of life (Adams 2005, Savard et al. 2000). However, overabundance of some urban wildlife populations can be undesirable and will require effective management strategies for the inevitable human/wildlife conflict, public safety concerns, or nuisance behavior (Conover 1997, Adams and Lindsey 2005).

An important first step in understanding the impact of rural to urban land use change is assessing possible differences in demographic parameters or habitat use of wildlife populations. The modification of vegetation cover or structural features in urban areas, including the possible alteration of predator community composition and availability of resources, will likely influence predation risks. Therefore, we require a greater understanding of urban ecology and how changes in habitat cover or predator communities may influence reproductive process, survival, or habitat use on urban landscapes.

This study examines the influence of urban land use on wild turkey (*Meleagris gallopavo*) behavior and survival. The wild turkey, like many avian species, is quite sensitive to changes in habitat structure and composition, thus making the species an excellent study candidate to investigate of the influence of land-use change. Specifically, my study goal is to investigate wild turkey demographics and home range use on three study areas of varying urban intensity in the Minneapolis-St. Paul, Minnesota, USA area. I achieve this goal by assessing the wild turkey's response to changes in land use, vegetation structure, disturbance, predation, and human presence. As habitat conditions encountered by urban wild turkeys likely differ from rural settings, it seems reasonable that these differences may impact nesting success, reproductive measures, survival rates, and home range characteristics. In addition, by examining the wild turkey's response on my urban study areas to published literature for the wild turkeys residing in rural habitats, we may gain further insights into the effects of urbanization.

In chapter 2, I investigate the influence of available vegetative cover and habitat features on wild turkey nesting success and reproductive measures in relationship to varying urban intensity. Urbanization profoundly alters existing habitat characteristics, creating markedly different habitats from rural or natural habitats (Marzluff and Ewing 2001, Shochat et al. 2006). For ground nesting birds, vegetation structure and composition for adequate concealment is paramount (Mankin and Warner 1992, Hagen et al. 2004, Kaczor 2008, Doherty et al. 2010). As a ground nesting species the wild turkey is no exception to these requirements, generally relying on nesting area features for

concealment and predator avoidance. Although the wild turkey is known to nest in a diverse set of cover conditions, females tend to select nest sites with greater concealment, including greater understory vegetation and canopy cover at one or more levels, with grassland sites typically near a shrub or other dense patch of vegetation (Day et al. 1991, Badyaev 1995, Thogmartin 1999). My goal is to assess whether changes associated with urban habitat cause a measured difference in wild turkey nesting success or reproductive measures. To achieve this goal, I examine nesting success as related to nesting land cover features and nesting measures in relationship to urban intensity.

In chapter 3, I assess wild turkey survival and cause-specific mortality in the urban setting. A common misconception concerning urban wildlife is that species are under less stress than their rural counterparts due to the presence of fewer predators and more abundant food resources (Gering and Blair 1999, Ditchkoff et al. 2006). However this view may be overly simplistic, and in reality, some wildlife species in urban areas are exposed to a novel array of stressors such as predator density changes (Harris 1977, Riley et al. 1998) and numerous sources of accidental human-caused mortality (Loss et al. 2012). My goal is to assess wild turkey survival and cause-specific mortality at sites characterized by different levels of urban intensity. To achieve this goal, I examine wild turkey survival and cause-specific mortality of females and poults, including survival during different biologically-relevant periods, and how these measures vary in relation to urban intensity.

In chapter 4, I investigate the influence of urban habitat on home range size and habitat use. Rural wild turkey populations are primarily influenced by two factors: predation and local resources, both related to habitat (Vangilder and Kurzejeski 1995, Roberts and Porter 1996). Yet despite the species perceived avoidance of human presence and variances in urban habitat, many wild turkey populations remain remarkably resilient in highly developed urban areas. My goal is to examine how changes to habitat and human disturbance influence the home range characteristics of the wild turkey in the urban setting. To achieve this goal, I assess home range size and habitat use in relationship to urban intensity.

The basis of this dissertation was not only to provide an understanding of urban wild turkey ecology, but to help inform urban conservation decisions regarding ground nesting birds. By clarifying the response of avian species to unique urban habitats, we gain a greater understanding of what provides suitable habitat for the birds and whether there are management recommendations that could result in improved habitat. In addition, while the creation or management of more 'natural' habitats in urban areas may relieve some urban pressures, this may lead to negative human-wildlife interactions which have occurred with some urban wild turkey populations. Therefore, the conclusions from this study can help guide us regarding the response of ground nesting bird species in the urban habitat, including helping to inform management strategies to improve ground nesting habitat within urban areas. Additionally, this study will assist urban wildlife managers to effectively manage wild turkey populations should conflicts arise.

Chapter 2: City living – the effects of an urban existence on Wild Turkey nesting success and reproductive ecology.

Introduction

Urbanization continues to occur at a rapid pace globally, altering natural areas that once consisted of prairie, woodland, or desert habitat into human-dominated habitats of pavement, buildings, and maintained lawns (Adams et al. 2005, Brown et al. 2005). In the lower continental United States, for example, human-developed (urban and suburban areas) land-use increased from approximately 6% to 27% between 1950 and the 1990s (Brown et al. 2005). Traditionally, conservation has centered on the use of protected areas (Soulé and Terborgh 1999), focusing heavily on wildlands, parklands, and other natural areas, while often overlooking 'natural-like' areas within our cities (Marzluff and Rodewald 2008). Of all forms of land-use change governed by human actions, none alter landscapes or influence wildlife to a greater degree than urbanization (McIntyre and Hobbs 1999, Czech et al. 2000, Marzluff and Ewing 2001, Chace and Walsh 2004). As our urban areas continue their relentless spread, the persistence of many species will likely be influenced by the successful incorporation of urban areas into the greater conservation context.

In the past few decades, several researchers have reviewed the state of urban wildlife research, often describing our understanding of the urban ecosystems as limited at best (Leedy 1979, Chace and Walsh 2004, Adams 2005). Research has demonstrated that human land-use can influence wildlife behavior, including altering population dynamics

and demographics (Theobald et al. 1997, Marzluff 2001, Hansen et al. 2005), even eliciting evolutionary responses (Badyaev 2005). Specifically, human land-use is associated with changes to the composition of local predator communities, modification of habitat features, disturbance, and spatial arrangement of key resources (Martin and Roper 1988, Newton 1993, Haskell et al. 2001, Marzluff 2001). To attain effective urban management, we require a basic knowledge of how wildlife responds to these changes in the urban setting, thus allowing for the full conservation value of human-developed areas to be realized.

The Wild Turkey (*Meleagris gallopavo*) is a prime example of a species that has successfully colonized an increasingly number of urban areas across its geographic range. The suitability of urban habitats is in sharp contrast to early Wild Turkey research expectations which described suitable habitat as absent of human presence and activities (Wright and Speake 1976) and consisting largely of forested areas (up to 25,000 acres [Mosby and Handley 1943]). Indeed, perceptions of suitable Wild Turkey habitat and species management has evolved greatly as conservation efforts not only reintroduced the species into prime Wild Turkey habitat, but also introduced the species into a greater variety of rural and agricultural landscapes. It is the Wild Turkey's ability to sustain viable populations in various habitats that provide us with an excellent opportunity to examine a species demographic response to differing habitats.

Traditionally, Wild Turkey research focused heavily on assessing habitat quality and suitability of rural and agricultural areas for species introduction or population management (Little and Varland 1981, Vander Haegen et al. 1988, Thomas and Litvaitis 1993, Wright et al. 1996, Thogmartin and Schaeffer 2000, Wright and Vangilder 2001, Hubert 2004, Wilson et al. 2005, Humberg et al. 2009). This research provides a wealth of information regarding the influence of various rural land uses and predator communities on Wild Turkey demographics and habitat use. Traditionally, Wild Turkey have been described as a secretive, ground nesting species dependent on nesting area vegetative features for concealment and predator avoidance. However, if habitat features available to urban Wild Turkey differ significantly from rural habitats, then these differences may lead to changes that are not representative of published rural reproductive measures or nest site use.

As many public agency wildlife budgets continue to constrict, urban wildlife management decisions may be based on rural demographics. However, we believe that this view may be overly simplistic, and in reality, some wildlife species in urban areas are exposed to a novel array of stressors such as changes in predator density (Harris 1977, Riley et al. 1998) and increases in disturbance from humans or domestic pets (Miller and Hobbs 2000, Loss et al. 2012). This calls into question the appropriateness of using results from rural research as the basis of wildlife management policy or conservation planning in urban areas.

In light of these novel stressors that may potentially affect the reproductive process, we conducted a Wild Turkey study at sites characterized by different levels of urban intensity in Minneapolis-St. Paul, Minnesota, USA. In this study we attempt to assess the influence of urban development on reproductive measures, nest success, and nesting behavior. Our objectives were: (1) to quantify the overall nesting performance of Wild Turkeys in relation to urbanization, and (2) to examine nesting success rates in relation to nesting area features.

Methods

Study Areas

The Lake Elmo rural-fringe (hereafter, "rural") study area was located in Washington County, Minnesota; the Snail Lake (suburban) and Battle Creek (urban) study areas were located in Ramsey County, Minnesota. The rural site consisted of agricultural areas (row crops and livestock operations), large tracts of mixed-use recreational parkland and natural areas, interspersed with mostly low-density residential areas. Parkland in the rural study area included Lake Elmo Park Reserve, a 2,165 acre mixed-use recreational area, and several large areas of maintained grassland and mixed hardwood stands owned by the Minnesota Department of Transportation; Minnesota Correctional Facility, Oak Park Heights, Minnesota; or the city of Bayport, Minnesota.

The suburban study area encompassed several county park units, including Snail Lake and Grass Lake Regional Parks, and the Arden Hills Army Training Site (AHATS). The

suburban study site was characterized by residential neighborhoods of various housing densities interspersed with parkland of varying recreational or military use. Civilian parkland ranged from high-use recreational areas (mowed and highly maintained) to low-use parkland managed for native plant species. AHATS was a 1,500 acre site that is leased by the Minnesota National Guard for training purposes. Large sections of AHATS were managed for native plant species, including prairie grassland and oak savannah species.

The urban study area included Battle Creek Regional Park, sections of the National Park Service's Mississippi River and Recreation Area, Minnesota Department of Natural Resources land, and city of St. Paul land. The urban site's parkland consisted mostly of high-use recreational areas, and adjacent to moderate to high density residential areas. This site contained small quantities of oak and mixed hardwood woodlands, wet lands, and grassland areas.

We observed several known Wild Turkey predators at all study areas, including raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), fox (*Vulpes vulpes*), coyotes (*Canis latrans*), domestic dogs (*Canis lupus familiaris*), domestic and feral cats (*Felis catus*), and several raptor species.

Capture and Monitoring

We captured Wild Turkeys from early December through late March between 2010 and 2012 using the accepted methods of air-netting, drop nets, and walk-in live traps (Glazener et al. 1964, Bailey et al. 1980, Gaunt et al. 1999, Nicholson et al. 2000). At capture, we classified each bird by sex and age-class defined as either juvenile (less than one year of age) or adult (after their first nesting season) based on feather and physical characteristics (Williams 1961, Brenneman 1992, Pelham and Dickson 1992, Schroeder and Robb 2005). We fitted all female birds with 78 g motion-sensitive VHF radio transmitters equipped with a 8 hour time delay mortality sensor (Advanced Telemetry Systems, Isanti, Minnesota) using a back-pack configuration (Roberts and Porter 1996, Wilson and Norman 1996, Norman et al. 1997). We handled and released all birds at the capture site according to an approved University of Minnesota Animal Research and Care Protocol (IACUC #0911A74374).

We used hand-held receivers and 3-element Yagi antennas to monitor survival and locate radio-tagged females at least three times per week during spring and summer (1 April to 30 September) and at least two times per week during autumn and winter (1 October to 31 March). We monitored for onset and termination of nesting activities through behavioral patterns, such as highly localized movements or mortality sensor activation (Porter 1978, Vander Haegen et al. 1988, Thogmartin 1999, Nguyen et al. 2004, Spohr et al. 2004). Once a nesting attempt was identified, we approximated nesting locations using radio signal strength and circling the female's location at a distance of at least 30 m to

minimize risk of disturbance, creating a scent trail, or accidental flushing (Gaunt et al. 1999, Thogmartin 1999, Nguyen et al. 2004).

We determined nest fate and clutch size by examining nest bowls for shell fragments, condition of fragments, and the presence of unhatched eggs (Porter 1983). We assigned each nest a fate of: (1) 'successful' if eggshells were cleanly broken and at least one egg hatched; (2) 'abandoned' if the female was not in the area on two consecutive visits and eggs were cold to the touch; or (3) 'depredated' if the eggs were smashed or missing (Hernandez et al. 1997, Nguyen et al. 2004). We recorded clutch size and number of eggs hatched when confident that an accurate and complete count could be obtained. For unsuccessful nesting attempts, we used telemetry data to infer date of clutch initiation (Badyaev 1995). For successful nesting attempts that could not accurately be dated from telemetry data, we established an approximate date for onset of incubation by back dating 28 days from the suspected hatch date. We established an approximate date for onset of nesting by adding an equal number of days based on the clutch size to the 28-day incubation period, as described by Schmutz and Braun (1989) and Badyaev and Faust (1996). For nests found during normal field activities, we considered the nest successful if hatchlings were observed in the immediate nesting area within three days of suspected hatching. We recorded nest site locations using a global positioning system (GPS; Trimble Pathfinder XPS) or by interpreting digital orthophotos (1 m resolution) and digitizing locations with a geographic information system (ArcGIS 10; Environmental Systems Research Institute, Redlands, CA).

Habitat Data

We recorded habitat measurements within one week of nesting termination to minimize phenological differences. We recorded vegetation structure and composition based on a 20 m diameter nest-centered plot. Plot design and methods follow Badyaev (1995), Day et al. (1991), and Nudds (1977) with some modifications as described here.

We estimated shrub and tree counts along two perpendicular, but randomly oriented transects, each two meters in width. We defined trees as woody plants greater than 2.5 cm diameter at breast height (dbh) and greater than 3 m in height; shrubs as woody plants between 0.25 m and 3 m in height. We estimated percent canopy cover directly above nest bowl and at four randomly assigned perimeter points 90 degrees apart at a height of 1 m, assigning each to a category of less than 25%, 26 to 50%, 51 to 75%, and greater than 75%. We measured understory height (cm) at nest bowl and at four randomly assigned perimeter points 90 degrees apart. To estimate percent visual obstruction, we used a vegetation profile board to assign vegetative features to a category of less than 2.5%, 2.5 to 25%, 26 to 50%, 51 to 75%, 76 to 95%, and greater than 95% at two height intervals (0-50 and 51-100 cm). From nest center, we measured distance to nearest road, distance to nearest actively used human structure, distance to the first large tree (> 30 cm dbh) and distance to open water by direct measurement or by overlaying nest coordinates and interpretation digital orthophotos using ArcGIS 10. All measured variables are defined in **Table 2.1** and **Table 2.2**.

Data Analysis

We did not include any females in the survival analysis if fatality occurred within 14 days of capture. We assumed mortality during this 14 day period was associated with captured-related stress or transmitter/harness complications (Nenno and Healy 1980, Roberts et al. 1995, Miller et al. 1996). If a monitored individual survived an annual cycle (April 1 to March 31), then we considered nesting activities commencing on April 1 as independent observations from the prior year. All data analyses were completed in Project R 2.15.2 (R Core Development Team 2012) or ArcGIS 10.

Reproductive Data Analysis

We used nonparametric approaches for most analyses because data samples were relatively small and not distributed normally after standard transformation. We compared clutch size, nesting date initiation, percent nesting, and number of eggs hatched per clutch among study areas using Kruskal-Wallis test methods and *t*-tests for age differences. If the test indicated a significant difference, we used pair-wise multiple comparisons between the samples at $\alpha = 0.05$ level of significance. We computed hatching success as the number of hatchlings divided by the number of eggs present in the nest at the time of hatching. We combined all clutches in each study area for analysis of first clutch initiation date, clutch size, and hatch rate.

We calculated nest survival estimates and 95% confidence intervals using the staggered-entry design of the Kaplan-Meier method (Kaplan and Meier 1958, Pollock et al. 1989, Nur et al. 2004). While commonly used for radio-transmitter data (Pollock et al. 1989, White and Garrott 1990, Millspaugh and Marzluff 2001), Aldridge and Brigham (2001) also demonstrated its usefulness for nest survival. We based the nest survival period on an average 28-day incubation period (Mosby and Handley 1943). We used logrank tests (Pollock et al. 1989) to assess the test hypothesis that the estimated survival functions (i.e., by study area or female age-class) statistically differ.

Habitat Data Analysis

We constructed candidate models using biologically relevant combinations of nest habitat and landscape variables based on previous Wild Turkey research (Badyaev 1995, Day et al. 1991) and variables we hypothesized might be important in the urban environment. We evaluated habitat variables with logistic regression modeling (function 'glm' in Project R 2.15.2). We included study area (SITE) as a fixed effect in each candidate model to test for variability among study areas (Hosmer and Lemeshow 2000). We limited our habitat analyses to three *a priori* model sets, each examining the effect of the selected habitat variables at a different spatial scale (nest site, nest patch, and landscape). We used a theoretic approach to model selection based on Akaike's Information Criterion with a finite sample size correction (AIC_c) to rank models by degrees of support (Burnham and Anderson 2002). We considered the model with the lowest AIC*c* to be the best supported by the data (Burnham and Anderson 2002). We considered models within

two AIC_c points of the top model as competitors for which we computed Akaike weights (w_i) to provide weights of evidence in support of each model (Burnham and Anderson 2002). If the 95% confidence interval of a variable's odds ratio included 1, we deemed that variable to be uninformative (Hosmer and Lemeshow 2000). We built a final global model using the set of all plausible predictors indicated in top and competitive models across all three spatial scales. Predictors identified in the final global model set were considered as plausible predictors in relationship to nesting success. Lastly, we used the area under the receiver operating characteristic curve (ROC) to further evaluate the predictive accuracy of the model (Guisan and Zimmermann 2000, Hosmer and Lemeshow 2000, Pearce and Ferrier 2000).

Results

We monitored 60 female Wild Turkeys during the 2010 - 2013 nesting seasons. Forty-two nests were included in the success analyses; 44 nests were included in the habitat analyses. Across all study areas and nesting females, we observed a nesting success rate, including re-nest attempts, of 59% (n = 27) versus a 41% unsuccessful rate (n = 19). We attributed nesting failure to mammalian predation (n = 7), abandonment (n = 4), avian predation (n = 2), accidental flushing (n = 2), vehicle strikes (n = 1), weather (n = 1), and unknown (n = 2). Three nesting attempts were not included in the analyses because the female nested outside the study area.

Reproductive Process

Across all study areas and all female Wild Turkeys in 2010 - 2013, we observed an average first nesting rate of 73.7% (n = 57) (**Table 2.3**). Nesting rate for suburban females (86%) was higher than that of urban females (67%) or rural females (67%), but did not statistically differ (χ^2 = 2.44; DF = 2; P = 0.30). We found juvenile (81%) and adult (65%) nesting rates did not statistically differ (χ^2 = 1.67, DF = 1, P = 0.20).

Across all study areas and all female Wild Turkeys in 2010 - 2013, we observed an average onset of incubation date of May 2 (April 12 – June 19; n = 42) for first nesting attempts (**Table 2.4**). We found that adult females (April 26; n = 17) initiated incubation approximately 11 days earlier than juvenile (May 6; n = 25) during the 2010 - 2013 time period, which was statistically significant (t = -1.79, DF = 40, P = 0.04) (**Table 2.4**). We found suburban females (28 April, SD = 13.61, n = 18) initiated incubation approximately 6 days earlier than urban females (3 May, SD = 22.9, n = 10) and 8 days earlier than rural females (5 May, SD = 19.73, n = 14), however this difference was not statistically significant (F = 0.63, DF = 2, P = 0.54). The mean monthly temperature in March 2011 (-1.4 C) was lower than in 2012 (9.1 C) or 2010 (5.0 C).

Across all study areas and all female Wild Turkeys in 2010 - 2013, we observed an average clutch size of 10.2 eggs per nest (9 – 13; n = 42) (**Table 2.5**). We found that clutch size for rural females (10.86; n = 14) was larger than for suburban females (10.05; n = 19) or urban females (9.56; n = 9), and differed statistically (χ^2 = 8.30, DF= 2, P =

0.02, n = 42 nests). A pairwise comparison of groups determined clutch size of urban females differed significantly from the rural females (P = 0.02), but suburban and rural females (P = 0.27) and urban and suburban females (P = 0.70) did not statistically differ.

Across all study areas and all female Wild Turkeys in 2010 - 2013, we observed a hatch rate of 84% (n = 26) (**Table 2.6**). We found hatch rates for rural females (86%, n = 9) was higher than that of urban females (85%, n = 5) and suburban females (81%, n = 12), however this difference was not statistically significant (χ^2 = 3.63, DF = 2, P = 0.17, n = 26).

Across all study areas and all female Wild Turkeys in 2010-2013, we observed a nest survival rate of 0.56 (CI = 0.43 – 0.73; n = 42) (**Table 2.7**). Nest survival for rural females (0.60; CI = 0.40 – 0.91; n = 14) was higher than for suburban females (0.53; CI = 0.34 – 0.81; n = 18) and urban females (0.46; CI = 0.24 – 0.87; n = 10), but did not differ by study area (χ^2 = 0.20, DF = 2, P = 0.91). Nest survival for adult females (0.56; CI = 0.37 – 0.84; n = 17) and juvenile females (0.54; CI = 0.38 – 0.77; n = 25) did not differ among study areas (χ^2 = 0.00; DF = 1; P = 0.95).

Habitat Factors Related to Nesting Success

For nest success based on habitat variables at the nest site (3 m diameter on nest center), we had one model (NEST II) that was competitive ($AIC_c \leq 2$) with the top model (NEST I) (**Table 2.8**). We found the most predictive habitat variables in the top nest site models

were canopy cover directly above the nest bowl (OVERC), visual obstruction at the nest bowl ([PBLC]; 0 – 50 cm), understory height surrounding the nest bowl (UNDERC), and number of trees within 1.5 m of the nest bowl (TREEC). The top model (NEST I) had moderate support (w_i = 0.58) over the other competitive model (NEST II, w_i = 0.38).

The confidence interval (CI) for the estimated odds ratio for the TREEC variable in the NEST II model included 1, indicating this variable is not well supported (**Table 2.9**). We found the statistically supported variables included OVERC (NEST I, P = 0.021; NEST II, P = 0.014), PBLC (NEST I P = 0.006, NEST II P = 0.004), and UNDERC (NEST I, P = 0.040; NEST II, P = 0.036). We found that the ROC scores produced for the NEST I model ([OVERC + UNDERC + PBLC]; ROC = 0.66) and NEST II model ([OVERC + UNDERC + PBLC + TREEC]; ROC = 0.66) did not differ significantly (P = 0.99; replications = 10,000), which suggests the variable TREEC may not be important in discriminating the likelihood of nest success. We found that the ROC scores produced for the NEST Full model ([OVERC + UNDERC + PBLC + PLUC + TREEC + SHRUBC]; ROC = 0.66) and NEST I and NEST II models did not differ significantly (both P = 0.86; replications = 10,000). We found nest success was negatively related to canopy cover directly over the nest bowl and understory height surrounding the nest bowl, and positively related to visual obstruction (0-50 cm) at the nest site. Inclusion of the SITE variable did not produce significant results for either the full or top models.

For nest success based on habitat variables at the nest patch (20 m diameter on nest center), we determined a single model to be best (PATCH I) (**Table 2.8**). We found the most predictive variables in the top nest patch model were visual obstruction at mid-patch (PBLP, 0 – 50 cm; $P = 0.041$), and understory height at the perimeter (UNDERP; $P = 0.002$). While statistically supported, we found that the ROC scores produced for the PATCH Full model ([OVERP + UNDERP + PBLP + PBUP + SHRUBP + TREEP]; ROC = 0.60) and PATCH I model ([UNDERP + PBLP]; ROC = 0.61) did not differ significantly ($P = 0.99$; replications = 10,000), which suggests the variables UNDERP and PBLP may be important in discriminating the likelihood of nest success. We found nest success was negatively related to understory height at the perimeter and positively related to visual obstruction (0-50 cm). Inclusion of the SITE variable did not produce significant results for either the full or top model.

For nest success based on variables at the landscape scale, we determined a single model as best (LAND I) (**Table 2.8**). We found the most predictive variables in the top landscape model included distance to open water from nest center (WATER; $P = 0.065$) and distance to the first large tree (LGTREE; $P = 0.087$) although neither variable was statistically supported (**Table 2.9**). The CI for the estimated odds ratio for the WATER variable in the LAND I model included 1, further indicating this variable may not be well supported (**Table 2.9**). We found that the ROC scores produced for the LAND Full model ([SITE + ROAD + WATER + STRUCTURE + LGTREE]; ROC = 0.62) and LAND I model ([WATER + LGTREE]; ROC = 0.74) did not differ significantly ($P =$

0.56; replications = 10,000), which suggests the variables WATER and LGTREE may be important in discriminating the likelihood of nest success. We found nest success was positively related to distance to water and distance to a large tree. Inclusion of the SITE variable did not produce significant results for either the full or top model.

We created a GLOBAL model including all top model variables, and determined a single model as best (GLOBAL I) (**Table 2.8**). We found the most predictive habitat variables in the top global model were canopy cover directly above nest bowl (OVERC), visual obstruction at the nest bowl ([PBLC]; 0 – 50 cm), understory height surrounding the nest bowl (UNDERC), and distance to open water from nest center (WATER).

The CI for the estimated odds ratio for OVERC in GLOBAL I included 1, indicating this habitat variable is not well supported as a predictive variable for successful versus unsuccessful nesting outcomes (**Table 2.9**). The variables with statistically significant support were UNDERC (GLOBAL I, $P = 0.037$), PBLC (GLOBAL I, $P = 0.003$), and WATER (GLOBAL I, $P = 0.024$). We found that the ROC scores produced for the GLOBAL Full model ([OVERC + UNDERC + PBLC + TREEC + UNDERP + PBLP + LGTREE + WATER]; ROC = 0.77) and GLOBAL I model ([OVERC + UNDERC + PBLC + WATER]; ROC = 0.73) did not differ significantly ($P = 0.23$; replications = 10,000), which suggests the variables OVERC, UNDERC, PBLC, and WATER may be important in discriminating the likelihood of nest success. We found nest success was positively related to visual obstruction (0 – 50 cm) at nest center and distance to open

water from nest center, and negatively related to understory height surrounding the nest bowl and canopy cover directly above nest bowl. Inclusion of the SITE variable did not produce significant results for either the full or top model.

Discussion

Urbanization is continuing to occur at a rapid pace, transforming natural habitat, such as forests, prairies, and wet lands with human dominated habitats of pavement, buildings, and maintained lawns (Adams et al. 2005, Brown et al. 2005). As urban land use continues to increase, we need a greater understanding of how species respond to urban habitats, including how these unique habitats influence demographic patterns and behaviors.

Numerous rural and agricultural studies have examined the Wild Turkey's reproductive process and nest habitat use; however, we lack information on how urban habitat influences these behaviors. Regardless of rural habitat type occupied, it is accepted Wild Turkey nest selection is not a random a process (Badyaev 1995, Thogmartin 1999). Instead nest site features are selected for visual and olfactory concealment from predators, a major source of nest failure across the species range (Speake 1980, Humberg et al. 2009).

For this study, we assessed several reproductive measures for the Wild Turkey in relation to urban intensity. Overall, we found the only Wild Turkey nesting measure that differed

significantly in relation to urbanization was clutch size. We also found habitat features most predictive of nest success were understory obstruction at the nest bowl, height of vegetation at the nest bowl, and distance to open water from nest center.

The Reproductive Process

We found a first nest-initiation rate of 73.7% across all females and study areas combined, which was lower, but within the range of nest-initiation rates reported for less urbanized areas (Van Haegen et al. 1988, Vangilder 1992, Vangilder and Kurzejeski 1995, Hubert 2004), and for a suburban Connecticut population (Spohr et al. 2004). Our lower observed nesting rate was likely influenced by weather patterns during the 2010 winter and 2011 nesting seasons. The 2010 winter (December through March) and 2011 early nesting seasons (April) experienced higher precipitation and cooler temperatures with a snow pack that did not fully melt in our study areas until mid-April (Minnesota Climatology Working Group, National Weather Service). We hypothesize that these climatic conditions likely influenced normal nesting patterns. Because of adverse weather, fewer females may have attempted to nest. Although we did observe a higher overall nest initiation rate among juveniles, which suggests the importance of this population segment to the reproductive output of local populations (Blankenship 1992).

We found that across all females and all study areas, adult females initiated nesting activities significantly earlier than juveniles. Although we failed to detect a significant difference in the date of incubation initiation by study area, limited nesting habitat may

provide an explanation for juveniles initiating nesting activities later than adults. If the more experienced adults secure prime nest locations earlier due to nest site fidelity or familiarity with local area habitat characteristics, then juveniles may travel further or sample more habitats types to find suitable nesting areas. We observed this trend as especially pronounced in the urban study population, where adults initiated incubation approximately 17 days early than juveniles. Furthermore, we observed a high rate of emigration by females out of our urban study area, suggesting a possible lack of suitable nesting habitat.

We found first clutch size across all females and study areas was similar to previous reports from across the species range (Porter 1978, Vangilder and Kurzejeski 1995, Spohr et al. 2004). Our results indicate clutch size for urban females differed significantly from rural females, with suburban clutches intermediate in size. The decline in egg production between urban and rural females could represent a response to predation risks (Zanette et al. 2006, Van Kleef et al. 2007) or poor food resources for nestlings (Chamberlain et al. 2009). Our observed reduction of clutch size in relation to degree of urbanization could reflect the influence of changing predator communities in urban areas. With a potentially higher rate of nest predation, urban females may find it favorable to reserve resources for future nesting attempts or to have fewer poults during the early brood rearing period. While we did not attempt to quantify available food resource for poults, most urban and suburban females' brood territory included areas with the potential for insect resources (i.e., gardens, parkland, planted prairies, non-maintained grassy road side areas).

We found nest survival rates across all females and study areas were similar to previously reported Wild Turkey nests in forested or agricultural areas (Treiterer 1987, Pringle 1988, Van Haegen et al. 1988) or in a suburban Connecticut population (Spohr et al. 2004). We found no evidence for differences in first nest survival rates by study area or by turkey age. Given the considerable variability in habitat and land use among the study areas, we found this somewhat surprising.

Habitat Variables Related to Nest Success

The importance of vegetation structure and composition in regards to concealment is important to ground nesting birds (Mankin and Warner 1992, Hagen et al. 2004, Kaczor 2008, Doherty et al. 2010). Urbanization leads to a profound restructuring of habitats, differing markedly from rural or natural habitats (Marzluff and Ewing 2001, Shochat et al. 2006). Although the Wild Turkey uses a variety of rural habitats to nest, nest site selection is not random (Lazarus and Porter 1985). Wild turkeys tend to select nest sites with greater concealment than the surrounding area, including a high degree of understory vegetative density and more open midstory and canopy cover (Lazarus and Porter 1985, Holbrook et al. 1987, Badyaev 1995, Godfrey and Norman 2001). Wild Turkey grassland nest sites are typically near a shrub or other dense patch of vegetation (Day et al. 1991, Porter 1992, Badyaev 1995, Thogmartin 1999).

We observed similar characteristics with our monitored nest sites typically exhibiting high visual concealment at the nest bowl, and declining concealment away from the nest. Overall, we found Wild Turkey nest success showed a strong trend for habitat variables related to vegetative density and height at the nest bowl scale, and distance to open water on the greater landscape. Important habitat predictors included in the top GLOBAL model suggests visual obstruction (0-50 cm) at the nest bowl and distance to open water are positively correlated to success, whereas average understory height at nest center and overhead cover density directly above the nest bowl are negatively correlated to success.

We found a higher degree of ground level visual obstruction at the nest bowl predictive of successful nests. Our results were similar to previous research which suggests Wild Turkey select nesting cover by understory features, including visual obstruction between ground level and 1 m at the nest site (Holbrook et al. 1987, Wertz and Flake 1988, Day et al. 1991, Badyaev 1995, Godfrey and Norman 2001, Hubert 2004, Nguyen et al. 2004). Nest site features, such as vegetative density and overhead cover, are presumably selected to provide visual, olfactory, and physical barriers to predators at the nest site (Martin 1993).

The presence of a tree or shrub adjacent to the nest bowl is a commonly reported feature reported in Wild Turkey nesting literature (Porter 1992). However we did not observe this trend for our urban or suburban birds. Conversely, most successful nests were found in grassy areas that can be best described by the lack of lateral. While successful females

appear to select nesting areas with a higher degree of vegetative obstruction at the nest bowl, it is unclear why urban females in particular selected sites with lower lateral cover. Lateral cover is likely part of a complex interaction between available habitat features and predation pressures. However, this behavior may be a response to shifts in avian predation risks in the urban environment.

While nests are often associated with vegetative features sufficient to conceal the nest, understory characteristics need to optimize visibility surrounding the nest from ground level (Logan 1973, Speake et al. 1975, Holbrook et al. 1987). For this study, average understory height tended to decline from nest center towards the nest patch perimeter. This may indicate an advantage to having a greater decline in vegetative height as one moves from the nest center is an increase in the field of vision surrounding the nest.

Lastly, we found successful nests were, on average, located at a significantly greater distance from water sources (148 m) than unsuccessful (89 m) nests. Prior research has been contradictory regarding the importance of water to Wild Turkey nesting success. For example, our results agree with Badyaev (1995) and Miller et al (1999), but contrary to observations reported by Nyugen et al (2004) and Thogmartin (1999). One plausible explanation successful nests were located a greater distance from open water is the greater density of mesopedators using habitat near open water. One such ubiquitous mesopredator observed on all of our study sites was the raccoon, which Speake (1980) and Thogmartin (1999) reported as a common Wild Turkey nest predator. Several studies

have reported availability of open water can influence raccoon distribution and abundance (Stuewer 1943, Sanderson 1987, Gerht and Fritzell 1998, Henner et al. 2004).

Conclusion

In the past several decades, research has increasingly focused on the response of wildlife to urbanization, providing evidence that species in human-modified areas are changing their behaviors. At present, urban wildlife managers are confronted with a complex array of decisions including not only species management, but also the role urban areas may have in local and regional conservation actions. The Wild Turkey has successfully colonized many urbanized areas throughout it range, producing viable populations in a habitat once thought to be unsuitable for the species. Early Wild Turkey research suggested that human contact was detrimental (Mosby and Handley 1943, Latham 1956), with Davis (1976) noting that poor turkey habitat includes areas with dense human populations or high levels of human activity.

Understanding the reproductive process of wildlife species occurring in urbanized habitats, and the habitat use in urban habitats, are important to urban conservation planning. Results of this study suggest that reproductive processes and habitat use of reproductively active female Wild Turkey differs in relationship to urbanization. It was beyond the scope of this study to determine whether specific factors (e.g., predator community changes, resources) changed with increasing urbanization. However, our results suggest we should not assume reproductive processes and habitat use applicable to

rural populations are representative of urban populations. We suggest future research of urban wildlife is needed to better understand possible responses to urbanization, to identify ecological and anthropogenic factors influencing these changes.

Tables

Table 2.1. Measured habitat variables grouped into three *a-priori* model sets for evaluating the variables influence on nesting success at three spatial scales by Wild Turkeys in east-central Minnesota, 2010- 2012.

Variable Name	Description
Nest Bowl Candidate Model Variable Names	
UNDERC	Average height of vegetation within 1.5 m of the nest bowl (cm)
OVERC	Estimated canopy cover directly above the nest bowl (category)
SHRUBC	Average number of shrubs within a 1.5 m of nest bowl center (#)
TREEC	Average number of trees within a 1.5 m of nest bowl center (#)
PBLC	Visual obstruction from 0-50 cm at nest center (%)
PBUC	Visual obstruction from 51-100 cm at nest center (%)
SITE	Study area indicator
Nest Patch Candidate Model Variable Names	
UNDERP	Average height of vegetation at the nest patch perimeter (cm)
OVERCP	Average canopy cover at the nest patch perimeter
SHRUBP	Average number of shrubs located along two 3 m wide transects the length of the nest patch (#)
TREEP	Average number of trees located along two 3 m wide transects the length of the nest patch (#)
PBLP	Visual obstruction from 0-50 cm 5 m from nest bowl center (mid-

Variable Name	Description
	patch) (%)
PBUP	Visual obstruction from 51-100 cm 5 m from nest bowl center (mid-patch) (%)
SITE	Study area indicator
Landscape Candidate Model Variable Names	
LGTREE	Distance from nest bowl to the first large tree (>30 cm dbh) (m)
ROAD	Distance from nest bowl to the nearest road (≥ 2 lanes) (m)
WATER	Distance from nest bowl to open water (m)
STRUCTURE	Distance from nest bowl to an occupied structure (m)
SITE	Study area indicator

Table 2.2. Averages for habitat variables measured at 44 Wild Turkey nest locations in east-central Minnesota, 2010-2012.

Successful and Unsuccessful Nest Habitat Variable Values							
Variable	Successful				Unsuccessful		
	Urban	Suburban	Rural		Urban	Suburban	Rural
UNDERC	75.2	57.2	74.8		65.2	60.1	40.8
UNDERP	42.8	47.3	56.0		50.2	50.2	39.0
OVERC	1.0	2.1	2.1		2.6	2.8	3.25
OVERP	1.2	1.3	1.4		2.0	1.2	1.8
SHRUBC	0.0	4.0	1.4		0.0	4.1	0.3

Successful and Unsuccessful Nest Habitat Variable Values						
Variable	Successful				Unsuccessful	
	Urban	Suburban	Rural		Urban	Suburban	Rural
SHURBP	2.4	29.2	41.5		2.6	42.5	52.0
TREEC	0.0	0.9	0.3		0.2	0.4	2.8
TREEP	0.4	5.9	7.3		0.8	4.7	9.5
Distance Measurements						
LGTREE	16.6	27.7	32.9		16.8	6.4	1.9
ROAD	65.4	55.8	134.5		94.2	57.6	232.3
WATER	172.2	155.4	127.7		128.2	60.4	113.2
STRUCTURE	154.8	250.2	201.4		253.4	232.5	252.0
Visual Obstruction, category						
PBLC	4.8	5.1	5.7		5.1	4.2	4.1
PBUC	2.2	2.9	3.5		3.2	2.5	1.9
PBLP	4.7	4.9	5.4		5.0	3.7	3.8
PBUC	2.3	2.6	3.0		3.0	2.1	1.4

Table 2.3. Number of female Wild Turkey alive at the beginning of nesting season, number of hens that attempted to nest and nesting percentage for three study areas in east-central Minnesota, 2010-2012. (Site R = Rural, Site S = Suburban, Site U = Urban).

		Adult			Juvenile			Total		
Site	Year	#	#	%	#	#	%	#	#	%

		Hens	Nest	Nest	Hens	Nest	Nest	Hens	Nest	Nest
R	2010	--	--	--	1	1	100	1	1	100
R	2011	4	2	50	6	3	50	10	5	50
R	2012	6	4	67	4	4	100	10	8	80
R	2010 - 2012	10	6	60	11	8	73	21	14	67
S	2010	1	1	100	0	0	0.0	1	1	100
S	2011	2	2	100	4	4	100	6	6	100
S	2012	5	4	80	9	7	78	14	11	79
S	2010 - 2012	8	7	88	13	11	85	21	18	86
U	2010	0	0	0.0	1	0	0.0	1	0	0.0
U	2011	3	1	33	3	3	100	6	4	67
U	2012	5	3	60	3	3	100	8	6	75
U	2010 - 2012	8	4	50	7	6	86	15	10	67

Table 2.4. Average and range of first nest incubation initiation date observed at 42 female Wild Turkey nest locations in east-central Minnesota, 2010-2012. (Site R = Rural, Site S = Suburban 2, Site U = Urban).

Site	Year	Adult	Juvenile	All
R	2010-2012	2 May (Apr 20-May 29; *n* = 6)	7 May (Apr 19-Jun 19; *n* = 8)	5 May (Apr 19-Jun 19; *n* = 14)
S	2010-2012	23 Apr (Apr 13-Jun 1; *n* = 7)	2 May (Apr 14-May 28; *n* = 11)	28 Apr (Apr 13-Jun 1; *n* = 18)
U	2011-2012	23 Apr (Apr 16-Apr 30; *n* = 4)	10 May (Apr 12-Jun 19; *n* = 6)	3 May (Apr 12-Jun 19; *n* = 10)

Table 2.5. Number of nests monitored and clutch size for female Wild Turkeys nests in east-central Minnesota, 2010-2012 (Mean ±1 SD).

	Rural			Suburban			Urban		
	# of Nest	Mean ± SD	Range	# of Nest	Mean ± SD	Range	# of Nest	Mean ± SD	Range
2010	1	10.0	10	1	10.0	10	--	--	--
2011	5	10.0 ± 0.71	9 - 11	6	9.3 ± 0.52	9 - 10	4	9.2 ± 0.50	9 - 10
2012	8	11.5 ± 1.20	10 - 13	12	10.4 ± 0.79	9 - 12	5	9.8 ± 0.84	9 - 11
Tota	14	10.86	9 - 13	19	10.05	9 - 12	9	9.56 ±	9 - 11

	Rural			Suburban			Urban		
	# of Nest	Mean ± SD	Range	# of Nest	Mean ± SD	Range	# of Nest	Mean ± SD	Range
l		± 1.16			± 0.85			0.73	

Table 2.6. Hatch rate for female Wild Turkeys clutches in east-central Minnesota, 2010-2012.

		Adult			Juvenile			Total		
Site	Year	# Eggs	# Hatch	% Hatch	# Eggs	# Hatch	% Hatch	# Eggs	# Hatch	% Hatch
R	2010	0	0	0	11	9	82	11	9	82
R	2011	0	0	0	31	27	87	31	27	87
R	2012	34	29	85	34	30	88	68	59	87
R	2010 - 2012	34	29	85	66	57	86	110	95	86
S	2010	10	9	90	0	0	0	10	9	90
S	2011	0	0	0	19	15	79	19	15	79
S	2012	43	33	77	39	32	82	82	65	79
S	2010 - 2012	53	42	79	58	47	81	111	89	81

		Adult			Juvenile			Total		
Site	Year	# Eggs	# Hatch	% Hatch	# Eggs	# Hatch	% Hatch	# Eggs	# Hatch	% Hatch
U	2010	--	--	--	--	--	--	--	--	--
U	2011	10	9	90	9	8	89	19	17	89
U	2012	10	8	80	20	16	80	30	24	80
U	2010 - 2012	20	17	85	29	24	83	49	41	85

Table 2.7. Average nest survival rate by study area and age-class observed at 42 female Wild Turkey nest locations in east-central Minnesota, 2010-2012.

	Year	Number of Nest	Survival Rate Estimate	95% CI (+/-)
Study Area				
Rural	2010-2012	14	0.60	0.91-0.40
Suburban	2010-2012	18	0.53	0.81-0.34
Urban	2011-2012	10	0.46	0.87-0.24
Age-class				
Adult	2010-2012	17	0.56	0.84-0.37
Juvenile	2010-2012	25	0.54	0.77-0.38

Table 2.8. Support for models predicting nest success on habitat variables recorded for 44 Wild Turkey nests on Ramsey and Washington counties, Minnesota during the breeding seasons 2010-2012. Models are based on Akaike's information criterion corrected for small sample sizes (AIC*c*). We compared the different models with the null-hypothesis model that contained all habitat variables at three spatial levels (nest bowl, nest patch, landscape). A final hybrid variable set that included habitat variables from all top models evaluated their influence on nest success. K is the number of parameters in the model; ΔAIC*c* is the difference in AIC*c* between each model and the top model; Akaike weight (*wi*) is the weight of the evidence for model *i*; receiver operating characteristic (ROC) curve statistic is based on estimates of the area under the curve (AUC) and evaluates the discrimination of the model. See Table 2.1 for definition of model variables.

Model	*K*	AIC_c	ΔAIC_c	w_i	*ROC*
Nest Bowl Full Model and Competitive Top Models					
NEST FULL [OVERC + UNDERC + PBLC + PBUC + SHRUBC + TREEC]	6	59.12	5.60	0.04	0.66[a]
NEST II [OVERC + UNDERC + PBLC + TREEC]	4	54.34	0.82	0.38	0.66[b]
NEST I [OVERC + UNDERC + PBLC]	3	53.50	0.00	0.58	0.66
Full Nest Patch Model and Top Model					
PATCH FULL [OVERP + UNDERP + PBLP + PBUP + SHRUBP + TREEP]	6	63.95	10.21	0.01	0.60[a]
PATCH I [UNDERP + PBLP]	2	53.74	0.00	0.69	0.61

Model	*K*	AIC_c	ΔAIC_c	w_i	*ROC*
Full Landscape Model and Top Competitive Model					
LAND FULL [ROAD + WATER + STRUCTURE + LGTREE]	5	60.48	4.61	0.060	0.62[a]
LAND I [WATER + LGTREE]	2	55.87	0.00	0.620	0.74
Full Global Model and Top Competitive Model					
GLOBAL FULL [OVERC + UNDERC + PBLC + TREEC + UNDERP + PBLP + LGTREE + WATER]	7	58.19	9.13	0.01	0.77[a]
GLOBAL I [OVERC + UNDERC + PBLC + WATER]	4	49.06	0.00	0.57	0.73

[a]*ROC* curves for the full and top competitive model do not differ at a 0.05 level using Delong and bootstrap method (boot *n* = 10000)
[b]*ROC* curves among top competitive models do not differ at a 0.05 level using Delong and bootstrap method (boot *n* = 10000)

Table 2.9. Parameter and odds ratio estimates, including 95% confidence intervals (CI), for habitat variables included in top model(s) of 44 Wild Turkey nest on Ramsey and Washington counties, Minnesota during the breeding seasons 2010-2012. Habitat variables were evaluated at three scales: nest bowl, nest patch, and landscape.

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Top Nest Bowl Models					
NEST I					

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Intercept	-3.054	-7.457, 0.821	0.136	0.047	0.001, 2.272
OVERC	-0.772	-1.490, -0.154	0.021	0.462	0.225, 0.857
PBLC	1.592	0.562, 2.894	0.006	4.912	1.754, 18.061
UNDERC	-0.041	-0.087, -0.006	0.040	0.960	0.917, 0.994
NEST II					
INTERCEPT	-3.525	-8.332, 0.514	0.107	0.029	0.001, 1.673
OVERC	-1.097	-2.096, -0.303	0.014	0.334	0.113, 0.738
PBLC	1.746	0.668, 3.109	0.004	5.731	1.950, 22.397
TREEC	1.432	-0.664, 3.911	0.211	4.186	0.515, 19.970
UNDERC	-0.041	-0.086, -0.006	0.036	0.960	0.917, 0.994
Top Patch Model					
PATCH I					
Intercept	-4.595	-8.328, -1.549	0.001	0.010	0.001, 0.212
UNDERP	-0.035	-0.072, -0.003	0.041	0.966	0.931, 0.997
PBLP	1.451	0.607, 2.517	0.002	4.267	1.835, 12.397
Top Landscape Models					
LAND I					
Intercept	-1.210	-2.550, -0.057	0.052	0.298	0.078, 0.944
WATER	0.007	-0.001, 0.015	0.065	1.007	0.998, 1.015
LGTREE	0.051	0.009, 0.116	0.087	1.052	1.009, 1.123

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Top Global Models					
GLOBAL I					
Intercept	-6.753	-13.957, -1.472	0.029	0.001	-0.001, 0.229
OVERC	-0.717	-1.477, -0.059	0.063	0.488	0.228, 1.943
WATER	0.011	0.003, 0.022	0.024	1.011	1.026, 1.227
UNDERC	-0.046	-0.097, -0.008	0.037	0.955	0.907, 0.992
PBLC	2.111	0.899, 3.742	0.003	8.254	2.456, 42.169

Chapter 3: Survival and Cause-specific Mortality of Female and Poult Wild Turkeys across Urban, Suburban, and Rural Habitats in Central Minnesota.

Introduction

Over the past 50 years, human population growth and expansion have proceeded at an unprecedented rate, impacting up to half of the planet's surface via some form of human activity (Vitousek et al. 1997, Brown et al. 2005). A major component of human-caused global change is large-scale land cover change associated with intensified human uses (Turner et al. 1995). Traditionally, wildlife conservation and management has addressed the issue of land cover change by conserving biodiversity in wilderness areas, parks, and other natural areas (Marzluff and Rodewald 2008) and overlooking more human-dominated environments. Of all forms of land use change governed by human actions, none alter natural landscapes or influence wildlife to a greater degree than urbanization (McIntrye and Hobbs 1999, Czech et al. 2000, Marzluff and Ewing 2001). As urban land use increases, so does the interface and subsequent conflicts between humans and wildlife (e.g., coyote (*Canis latrans*) [Kellert 1985], Canada goose (*Branta canadensis*) [Conover and Chasko 1985]). As such, wildlife managers need to understand the conservation value of remaining 'natural' land in developed areas, and particularly, for species that may come into conflict with human populations.

Over the past several decades, several authors have reviewed the state of urban wildlife research (Leedy 1979, Chace and Walsh 2004, Adams et al. 2005, Marzluff and Rodewald 2008, McKinney 2008). Early urban wildlife research typically focused on

bird-habitat associations (for example, Pitelka 1942, Emlen 1974, Blair 1996); however, our understanding of basic urban ecology, including differences in demographic parameters of wildlife populations, such as reproductive rates, cause-specific mortality, and survival rates remains limited (Adams 2005). To more effectively manage urban wildlife populations, and to better understand human-wildlife conflicts, such as those caused by non-domestic species that rapidly increase their population abundance in urban areas, wildlife ecologists must understand population demographics of urban wildlife.

Wildlife in rural settings can provide immense economic and recreational benefits through such activities as sport hunting or wildlife viewing (Burger et al. 1999, Minnesota Department of Natural Resources 2007, U.S. Department of the Interior 2011). However, wildlife residing in urban settings, despite providing some similar benefits in urban parks and yards, are increasingly coming into conflict with humans, often posing considerable challenges to resource managers due to public safety concerns (e.g., vehicle-wildlife collisions and transmission of zoonotic diseases) or nuisance behavior (Conover 1997, Adams and Lindsey 2005, Minnesota Department of Natural Resources 2011). The wild turkey (*Meleagris gallopavo*) is a prime example of a species that has successfully colonized many urban areas across its geographic range. Early wild turkey researchers often described the species as intolerant of human presence (Wright and Speake 1976) and requiring large tracts of forested areas (up to 25,000 acres [Mosby and Handley 1943]). One such opinion summarized by Shaw (1959) articulates this point: "A wild turkey needs one thing for sure, lots of timberland and not much human

disturbance." However, the species has clearly expanded its range to include urban areas; in numerous metropolis areas, including Boston, Massachusetts, USA; Cleveland, Ohio, USA; and Minneapolis-St. Paul, Minnesota, USA, wild turkeys now commonly occur throughout the city, even in highly developed areas around urban central cores.

Most wild turkey survival studies have occurred in forested (Vander Haegen et al. 1988, Thomas and Litvaitis 1993, Thogmartin and Schaeffer 2000, Wright and Vangilder 2001, Wilson et al. 2005) or agricultural (Little and Varland 1981, Wright et al. 1996, Hubert 2004, Humberg et al. 2009) landscapes and may not be representative of wild turkey populations occupying urban areas. Habitat characteristics used by urban wild turkeys may differ from those used by rural turkeys and these differences may lead to differences in survival rates. For example, a common misconception concerning urban wildlife is that they are under less stress than their rural counterparts due to the presence of fewer predators and a more plentiful food supply (Gering and Blair 1999, Ditchkoff et al. 2006). However this view may be overly simplistic, and in reality, some wildlife species in urban areas are exposed to a novel array of stressors such as predator density changes (Harris 1977, Riley et al. 1998) and numerous sources of accidental human-caused mortality (Loss et al. 2012).

In light of these novel stressors that could potentially affect population dynamics, we conducted a study to assess survival and cause-specific mortality of wild turkeys at sites characterized by different levels of urbanization intensity in Minneapolis-St. Paul,

Minnesota, USA. Our objectives were: (1) to quantify overall survival and cause-specific mortality of female and poult wild turkeys, (2) to examine variation in female survival across different biologically relevant stages of the wild turkey annual cycle, as female survival rates may differ depending on seasonal activity periods, (Wunz and Hayden 1975, Vander Haegen et al. 1988, Miller et al. 1998), and (3) to examine how these measures varied in relation to urbanization.

Methods

Study Areas

The Lake Elmo rural-fringe (hereafter, "rural") study site was located in Washington County, Minnesota; the Snail Lake (suburban) and Battle Creek (urban) study sites were located in Ramsey County, Minnesota. The rural site served as a reference site because most of its area consisted of agricultural lands, large tracts of low-use recreational parkland and natural areas, and mostly low density residential areas. Parkland in the rural site included Lake Elmo Park Reserve, and several large areas of maintained grassland and mixed hardwood stands owned by the Minnesota Department of Transportation; Minnesota Correctional Facility, Oak Park Heights, Minnesota; or the city of Bayport, Minnesota.

The suburban site encompassed several county park units, including Snail Lake and Grass Lake Regional Parks, and the Arden Hills Army Training Site (AHATS). This study area was characterized by mostly residential neighborhoods interspersed with

parkland of varying recreational or military use. Civilian parkland ranges from high-use recreational areas (mowed lawns and highly maintained vegetation) to low-use parkland managed for native plant species. AHATS is a 1,500 acre site leased by the Minnesota National Guard for training purposes. Large sections of AHATS are managed for native plant species, including prairie grassland and oak savannah species.

The urban site included Battle Creek Regional Park, sections of the National Park Service's Mississippi River and Recreation Area, Minnesota Department of Natural Resources land, and City of St. Paul land. The urban site consisted mostly of high-use recreational parkland surrounded by dense residential neighborhoods. This site contained small quantities of oak and mixed hardwood woodlands, wet lands, and grassland areas.

We observed several known wild turkey predators at all study areas, including raccoons (*Procyon lotor*), stripped skunks (*Mephitis mephitis*), fox (*Vulpes vulpes*), coyotes (*Canis latrans*), domestic dogs (*Canis lupus familiaris*), domestic and feral cats (*Felis catus*), and several raptor species.

Capture and Monitoring

We captured wild turkeys from early December through late March between 2010 and 2012 using air-netting, drop nets, and walk-in live traps (Glazener et al. 1964, Bailey et al. 1980, Gaunt et al. 1999, Nicholson et al. 2000). At capture, we classified each bird by sex and age-class defined as either juvenile (less than one year of age) or adult (after their

first nesting season) based on body and feather characteristics (Williams 1961, Brenneman 1992, Pelham and Dickson 1992, Schroeder and Robb 2005). We fitted all female birds with 78 g motion-sensitive VHF radio transmitters (Advanced Telemetry Systems, Inc., Isanti, Minnesota) using a back-pack configuration (Roberts and Porter 1996, Wilson and Norman 1996, Norman et al. 1997). We handled and released all birds at the capture site according to an approved University of Minnesota Animal Research and Care Protocol (IACUC #0911A74374).

We used hand-held receivers and 3-element Yagi antennas to monitor survival and locate radio-tagged females at least three times per week during spring and summer (1 April to 30 September) and at least two times per week during autumn and winter (1 October to 31 March). We monitored for behavior that indicated the onset of nesting activities, nesting termination, and, if nests were successful, the onset of the brooding period (Porter 1978, Badyaev 1995, Godfrey and Norman 2001, Nguyen et al. 2004, Spohr et al. 2004). We established brood size based on evidence at the nest sites (i.e., examination of nest bowls for shell fragments, condition of fragments, the presence of disturbed/predated eggs, and visual confirmation with binoculars when permissible as not to disturb the poults) (Vangilder et al. 1987). We recorded brood size for radio-monitored females when confident that an accurate and complete count of poults could be obtained. Beginning at approximately 14 days post-hatch, we monitored poult survival on a weekly basis. We attempted to minimize brood disturbance using visual counts; however, where necessary to assess accurate counts, we completed flush counts (Vangilder et al. 1987,

Paisley et al. 1998). We terminated counts after four weeks post-hatch, if the brood or brood hen was lost, or if mixed broods formed.

We immediately investigated radio-equipped female mortality signals upon detection. We determined probable mortality cause using field signs and carcass condition, and if determinable, we classified the cause of mortality as mammalian predation, avian predation, vehicular collision, equipment failure (e.g., transmitter failure, harness malfunction, or entanglement), harvest, or unknown. We considered the source of mortality as predation when the carcass exhibited obvious evidence of trauma. Mammalian predation was identified by broken bones, shearing of wing feathers, bite mark evidence on bones and radio transmitter, or scattering of the remains over a large area (greater than 5 m^2) (Thogmartin and Schaeffer 2000, Elbroch 2003). In a substantial portion of cases we could not accurately differentiate mammalian predation to a specific entity, such as coyote, fox, or domestic pets; therefore we combined these mortalities into the broad category of mammalian predation.

We identified avian predation by carcass condition (removal of the head or neck, breast muscle removal, damage to quill section of feathers, or V-shaped notches in bones; (Smith 2002, Hardey et al. 2006), clean cut to the harness cord, or other avian species feathers in the general kill area. Vehicular collisions were considered the specific cause of mortality when the carcass exhibited blunt force trauma, such as crushed bones or

internal bleeding, no indication of consumption, and the carcass was located with 5 m of a road.

We attempted to minimize bias introduced from scavenging behavior by frequently monitoring for morality and investigating all mortality signals. Further, if the carcass showed signs of being moved (i.e., cached, no evident kill site in relation to the carcass location), or if the carcass showed signs of consumption for probable vehicular strikes, we assigned these cases as unknown cause of death. We attempted to differentiate scavenging from predation or vehicle collisions by freshness of the carcass, presence of hemorrhaging, relative pattern of consumption, and evidence of the carcass location in regards to the kill site (Elbroch 2003).

Data Analysis

We did not include any females in the survival analysis if fatality occurred within 14 days of capture. We assumed mortality during this 14 day period was associated with captured-related stress or transmitter/harness complications (Nenno and Healy 1980, Roberts et al. 1995, Miller et al. 1996). Further, we censored individuals that experienced radio transmitter failure, harness entanglement, or emigrated from the study area from the corresponding annual or seasonal analysis. When exact date of fatality was unknown, we assumed the mortality event occurred midway between previous and current monitoring dates. If a monitored individual survived an annual cycle (1 April to 31 March), then we

considered observations commencing on 1 April as independent observations from the previous year.

We calculated annual, seasonal, reproductive period, brooding female, and mammalian predation cause-specific survival estimates and 95% confidence intervals using the staggered-entry design of the Kaplan-Meier method (Kaplan and Meier 1958, Pollock et al. 1989, Nur et al. 2004). We generated annual survival estimates based on an annual cycle beginning 1 April and ending 31 March for each female by age-class and study area for the period of 2010-2013. We divided the annual cycle into 4 biologically relevant periods associated with seasonal activities occurring between 1 April through 30 June (spring/nesting), 1 July through 30 September (summer/brood rearing), 1 October through 31 December (autumn/flock build up), and 1 January through 31 March (winter/early breeding). We generated survival estimates for brooding females, which we defined as a period that started when females successfully hatched at least one poult and that lasted for 4 weeks post-hatch when several of the broods began to mix. We generated survival estimates for the incubation and early brood rearing period, which we defined as an average 28-day incubation period (Mosby and Handley 1943) and 28-day early brooding period, limited to reproductively active females associated with mammalian cause-specific mortality. We generated survival estimates for mammalian cause-specific mortality of females' first nesting attempts. We compared estimated survival curves using log rank tests (Pollock et al. 1989) for differences by study area and by female by age-

class. We conducted all statistical analyses in R 2.15.2 (R Core Development Team 2012).

Results

We monitored 60 female wild turkeys from 2010 to 2013. Thirty-seven mortalities were recorded from 1 April, 2010 to 1 April 2013. Five females were not included in analyses because they died within two weeks of capture, three females were censored because they left the study area, and an additional four females were censored or were seasonally censored because of transmitter failure (n = 3) or harness entanglement (n = 1).

Female Survival

Across all study areas and all female wild turkeys in 2010-2013, we observed an annual survival rate of 0.43 (CI = 0.32 – 0.58; n = 55) (Table 3.1). Annual survival rate for female adults (0.35; CI = 0.20 – 0.59; n = 25) and juveniles (0.48; CI = 0.34 – 0.70; n = 30) did not differ among study areas during the 2010-2013 time period (χ^2 = 0.7, DF = 1, P = 0.41). During 2010-2013, annual survival rate for suburban females (0.43; CI = 0.26 – 0.70; n = 20) was higher than for urban females (0.40; CI = 0.22 – 0.74; n = 14) and rural females (0.41; CI = 0.25 – 0.68; n = 21) birds, but did not differ significantly (χ^2 = 0.1; DF = 2; P = 0.97). Seasonal survival rate estimates for all females ranged from a low of 0.61 for the spring period to a high of 0.96 during the winter period (Table 3.1).

Across all study areas and all female wild turkeys in 2010-2013, we observed a spring survival rate of 0.61 (CI = 0.50 – 0.75; *n* = 55) (Table 3.1). Spring survival rates for female adults (0.58; CI = 0.42 – 0.80; *n* = 25) and juveniles (0.61; CI = 0.46 – 0.81; *n* = 30) did not differ among age-class during the 2010-2013 time period (χ^2 = 0; *DF* = 1; *P* = 0.84). During 2010-2013, spring survival for urban females (0.69; CI = 0.49 – 0.96; *n* = 14) was higher than for rural females (0.64; CI = 0.46 – 0.87; *n* = 21) and suburban females (0.48; CI = 0.30 – 0.75; *n* = 20), but did not differ significantly (χ^2 = 2.2; *DF* = 2; *P* = 0.33).

Across all study areas and all wild female turkeys in 2010-2013, we observed a summer survival rate of 0.83 (CI = 0.71 – 0.96; *n* = 34) (Table 3.1). Summer survival rates for female adults (0.75; CI = 0.57 – 0.99; *n* = 15) and juveniles (0.85; CI = 0.71 – 1.0; *n* = 19) did not differ among age-class during the 2010-2013 time period (χ^2 = 0.7; *DF* = 1; *P* = 0.42). During 2010-2013, summer survival rates for suburban females (0.90; CI = 0.73 – 1.0; *n* = 10) was higher than rural females (0.86; CI = 0.69 – 1.0; *n* = 14) or urban females (0.80; CI = 0.59 – 1.0; *n* = 10), but this difference was not statistically significant (χ^2 = 0.4; *DF* = 2; *P* = 0.82).

Across all study areas and all female wild turkeys in 2010-2013, we observed an autumn survival rate of 0.89 (CI = 0.75 – 0.99; *n* = 28) (Table 3.1). Autumn survival rates for female adults (0.83; CI = 0.65 – 1.0; *n* = 12) and juveniles (0.94; CI = 0.83 – 1.0; *n* = 16) did not differ among age-class during the 2010-2013 time period (χ^2 = 0.8; *DF* = 1; *P* =

0.39). During 2010-2013, autumn survival rates for suburban females (1.0; CI = 0.76 – 1.0; n = 9) was higher than rural females (0.91; CI = 0.75 – 1.0; n = 11) or urban females (0.75; CI = 0.50 – 1.0; n = 8), but this difference was not statistically significant (χ^2 = 2.8; DF = 2; P = 0.25).

Across all study areas and all female wild turkeys in 2010-2013, we observed a winter survival rate of 0.96 (CI = 0.89 – 1.0; n = 25) (Table 3.1). Winter survival rates for female adults (0.90; CI = 0.73 – 1.0; n = 10) and juveniles (1.0; n = 15) did not differ among age-class during the 2010-2013 time period (χ^2 = 1.5; DF = 1; P = 0.22). During 2010-2013, summer survival rates for suburban (1.0; n = 9) and urban females (1.0; n = 6) was higher than for rural females (0.90; CI = 0.73 – 1.0; n = 10), but this difference was not statistically significant (χ^2 = 1.5, DF = 2; P = 0.47).

We observed 42 females enter the reproductive period, which began with the onset of incubation and lasted up to 4-weeks post-hatch. Mammalian predation was the main cause of mortality and reproduction termination observed in this study, with mammalian predators taking 12 of 16 females during the incubation/early brood rearing period. As we intended to investigate mammal predation during the incubation/early brood rearing period, we did not include five females that abandoned their nesting attempt during the incubation period and an additional 6 females whose mortality agent was not mammalian cause-specific.

Across all study areas and all female wild turkeys in 2010-2013, we observed an incubation/early brood rearing survival rate of 0.59 (CI = 0.89 – 1.0; n = 31) (Table 3.1). Observed survival during this period was highest for rural females (0.67; n = 8), followed by suburban (0.53; n = 14) and urban (0.50; n = 9) females. Among the females analyzed, we did not detect a difference in overall female survival by study area (χ^2 = 1.381; P = 0.50) or age-class (z = 0.275; P = 0.78) during the reproductive period.

We generated a Kaplan-Meier survival curve to examine mammalian cause-specific mortality of female wild turkeys involved with and incubation/early brood rearing. The survival curve for females during their first nesting attempt was based on an average 28-day incubation and 28-day post-hatch period. Examination of Figure 1 indicates all observed mammalian cause-specific mortality occurred by day 35. Mammalian predation tended to be greatest the first week of the incubation period (41.7%) and the first week of the brood period (33.3%). We generated a Kaplan-Meier survival curve to examine timing of mortality of the females during incubation/early brood rearing period (Figure 2). We detected a difference in incubation/early brooding period survival (Figure 2) by study area (P = 0.01) but not by age-class (P = 0.97).

Poult Survival

During the brooding seasons of 2010 through 2012, we had accurate count data allowing for estimation of average poult survival for 26 broods (urban n = 5; suburban n = 11; rural n = 10). During the study period, an estimated 216 poults successfully hatched

(Table 3.2). Overall combined survival rate to 2 weeks post-hatch was 0.35, declining to 0.26 4-weeks post-hatch. Observed poult survival to 4 weeks was lower for female adults (0.24; n = 106) than juveniles (0.29; n = 110), but did not differ statistically (z = -0.886, P = 0.37). Observed poult survival to 4 weeks was highest for urban poults (0.31; n = 41), by suburban (0.25; n = 95) and rural (0.24; n = 80) poults, but did not differ statistically (χ^2 = 2.288, P = 0.32).

Sources of Mortality

Overall, mammalian and avian predation accounted for 63.3% of all observed female mortalities, followed by vehicle strikes (23.3%), harvest (3.3%), and unknown causes (10.0%) (Table 3.3). Predation remained the leading cause of mortality regardless of age-class, although predation tended to be higher in female adults (61.5%) than juveniles (47.1%). Most mammalian predation (75.0%) occurred during spring (1 April through 30 June) (Table 3.4). Deaths attributed to other factors were not as focused to one seasonal period.

For urban female wild turkeys, we found predation accounted for 85.7% of all recorded mortality, followed by vehicle strikes at 14.3% (Table 3.5). For suburban females, we found predation accounted for 45.5% of all recorded mortality, followed by vehicle strikes at 36.4%, and 18.2% attributed to unknown causes. For the rural females, we found predation accounted for 66.7% of all recorded mortality, followed by vehicular strikes at 16.7%, harvest at 8.3%, and 8.3% attributed to unknown causes. For the

incubation/early brood rearing period, predation accounted for 100% of urban fatalities, 87.5% of suburban fatalities, and 50.0% of rural fatalities.

Discussion

As urbanization continues to impact a larger proportion of the planet, there is a growing need to understand species responses. In the past several decades, research has increasingly focused on the demographic response of wildlife to urbanization, although much of this has proven to be inconclusive. For example, recent studies found nesting in urban areas was not detrimental to nesting success (Dykstra et al. 2000, Coleman et al. 2002), whereas others reported a reduction in fledging success (Boal and Mannan 1999). Here we examined wild turkey survival and cause-specific mortality in relation to urbanization. Overall, we found that predation was the primary source of mortality, which is consistent with previous research in more rural areas. Notably, however, we observed that a relatively large proportion of mortality was caused by collisions with vehicles.

We found that the annual wild turkey survival rate across all study areas combined tended to be lower than previously reported for female wild turkeys in less urbanized areas. Our results were lower than annual survival rates reported by other authors for mostly forested areas (Vander Haegen et al. 1988, Roberts et al. 1995, Miller et al. 1998, Wilson et al. 2005) or agricultural areas (Wright et al. 1996, Humberg et al. 2009). However, Hubert (2004) and Spohr et al (2004) reported similar survival estimates for an

agricultural and a suburban area, respectively. We found no evidence for differences in annual survival rates by study area or by turkey age. This result was somewhat surprising given the considerable variability in habitat and land-use among the study areas. The lack of differences in survival rates between age classes was likely a result of the juvenile cohort not joining the study until 6 to 9 months of age, and therefore already having survived the relatively high mortality typically associated with the nesting and brooding period.

We observed similar patterns of seasonal survival as reported for other regions in the United States, with female wild turkey survival typically lowest during the spring or reproductive period and higher but variable survival throughout the remainder of the year (Roberts et al. 1995, Wright et al. 1996, Miller et al. 1998, Hubert 2004). Our spring and reproductive period survival rates were lower than those reported by Humberg et al (2009) and Wright et al (1996) in mostly agricultural areas and by Roberts et al (1995) and Wilson et al (2005) in mostly forested areas. However, Spohr et al (2004) reported similar survival rates for a suburban population in Connecticut.

Our estimate of poult survival 4 weeks post-hatch was lower than estimates for 6 other studies summarized by Vangilder (1992). We also observed a high rate of complete brood loss during the 4 week post-hatch period. One potential benefit of inhabiting urban/suburban landscapes is that parkland and residential areas in these areas tended to have sparser undergrowth. The lack of undergrowth may have benefited local turkey

populations by providing a less obstructed view of their surroundings and therefore increased mammalian avoidance. Conversely, reduced undergrowth may influence avian predation or use of these areas by humans and domestic pets.

Similar to previous studies in rural landscapes (Vander Haegen et al. 1988, Roberts et al. 1995), we found female mortality primarily occurred during the spring or incubation and early brood rearing season. This is not surprising as this is a time when females spend most of their time on their ground nests (Williams et al. 1971, Green 1982) or constrained by pre-fledgling movements (approximately 2 weeks post-hatch). We observed mammalian predators, such as fox and coyote (Everett et al. 1980, Speake 1980, Wright et al. 1996), which presumably rely on olfactory cues to find nesting females or young broods. The limited movement of females during incubation and pre-fledgling brood rearing likely increased exposure risk of females and poults to mammalian predation.

Our results regarding female cause-specific mortality are consistent with findings from other wild turkey studies that reported predation as the leading cause of mortality (Treiterer 1987, Vangilder and Kurzejeski 1995, Vangilder 1996, Wright et al. 1996, Kubisiak et al. 2001). We found that 65% of spring mortality for all females, regardless of reproductive status, was attributable to predation (avian and mammalian), followed by 25% due to collisions with vehicles. By contrast, mortality of reproductively active females was 81% attributable to predation, with 75% attributable to mammalian predation alone, followed by vehicle collisions at 19%. Our results indicated that for

reproductively active females mortality caused by mammalian predation increased with increasing urbanization intensity. We are uncertain about the specific mechanisms responsible for this observed pattern; however, a potential explanation is one or more aspects of the local predator community or habitat changed with increasing urbanization.

Interestingly, we observed only one female mortality attributed to collision with a vehicle (8.3%) for the urban and suburban sites combined, whereas half of all mortalities in the rural study area were attributed to vehicle collisions during the reproductive period. The sole vehicle mortality for the urban/suburban sites likely occurred because accessing the closest water source to the nest site required crossing a busy 4-lane street. We are uncertain why the wild turkeys in the rural site experienced higher vehicle collision mortality, as we rarely observed these birds using or crossing roads other than sparsely-used park roads. This is in sharp contrast to suburban and urban birds, where frequent crossing of busy 4 lane roads and highways was observed. Combined with the observation that that urban/suburban birds rarely flew across roads, but rather walked or ran through, it is surprising that they did not experience higher rates of vehicle collision mortality.

To further investigate the impact of mammalian predation on reproductively active females, we restricted the survival analysis to only mammalian cause-specific predation from onset of incubation to 4-weeks post-hatch. We found female mortality was limited to the incubation and the first week post-hatch period. Our results indicated predation

timing differed by site with all mammalian predation occurring in the first two weeks of incubation in urban sites, during both periods in suburban sites, and late in the nesting cycle or early in the brooding period in rural sites. Again, it is unclear why there is such a difference in predation timing, but this difference may arise from differences among local predator community composition and patterns of daily and seasonal predator activity.

The survival rates reported in this study should be interpreted with caution due to the relatively short study duration (3 years; 2010-2013) and small sample size. Studies of longer duration have demonstrated that annual survival rates vary considerably within study areas. For instance, Vangilder and Kurzejeski (1995) reported annual turkey survival estimates in Wisconsin that ranged from 43% to 66% over a seven year period, and Wright et al. (1996) reported annual estimates ranging from 45% to 69% in Missouri over a 10 year period.

Management Implications

As an increasing number of "potentially problematic" wildlife species colonize or are introduced to the urban landscape, urban wildlife managers are confronted with complex management decisions. Managers must weigh both the advantages and dis-advantages (to both the wildlife and human populations) of managing to encourage or discourage wildlife populations in urban areas. Examples of human-wildlife conflicts due to increasing wildlife populations in the urban environment are increasingly common, with examples including deer-auto collisions, nuisance behaviors associated with Canada

goose presence, and conflicts between coyotes and domestic dogs. The wild turkey has successfully colonized many urbanized areas throughout it range, producing viable populations in a habitat once thought to be unsuitable for the species. While urban turkeys are generally viewed in a favorable light, nuisance behaviors, such as aggressiveness to humans and pets and scratching of vehicles associated with roosting behavior, and overabundance issues, has increased complaints to resource manages, both in the Twin Cities Metropolitan area and in other major cities of North America.

Understanding demographic rates of wildlife species occurring in urbanized habitats, and the factors affecting these rates, are important to urban wildlife managers. Results of this study suggest that mammal-caused mortality of reproductively active female and poult turkeys is elevated in highly urbanized settings. It was beyond the scope of this study to determine whether specific predator community factors (e.g., predator diversity and abundance) changed with increasing urbanization. However, our results suggest that wildlife managers should not assume that demographic rates that apply to rural populations also apply to urban populations of the same species. Additional research of urban and rural populations of wildlife species are needed to better understand species responses to urbanization, to identify ecological and anthropogenic factors that lead to unnaturally increased abundances of problem species, and to determine optimal management strategies that promote the presence of beneficial species and discourage the presence of nuisance species.

Tables and Figures

Table 3.1. Annual and seasonal Kaplan-Meier survival rate estimates (SE) pooled across years for radio-equipped female wild turkeys in east-central Minnesota, USA, for 2010-2013. (AN = Annual; SP = Spring; SU = Summer; AU = Autumn; WI = Winter)

	n	AN	*n*	SP	*n*	SU	*n*	AU	*n*	WI
Female (Combined)	55	0.43 (0.07)	55	0.61 (0.06)	34	0.83 (0.06)	28	0.89 (0.06)	25	0.96 (0.07)
Adult[a]	25	0.35 (0.09)	25	0.58 (0.10)	15	0.75 (0.11)	12	0.83 (0.11)	10	0.90 (0.09)
Juvenile[a]	30	0.48 (0.09)	30	0.61 (0.09)	19	0.85 (0.08)	16	0.94 (0.06)	15	1.00
Urban	14	0.40 (0.13)	14	0.69 (0.12)	10	0.80 (0.13)	8	0.75 (0.15)	6	1.00
Suburban	20	0.43 (0.11)	20	0.48 (0.11)	10	0.90 (0.09)	9	1.00	9	1.00
Rural	21	0.41 (0.10)	21	0.64 (0.10)	14	0.86 (0.09)	11	0.91 (0.09)	10	0.90 (0.09)

[a] Age-class is defined as Adult (≥1 year of age) or Juvenile (<1 year of age).

Table 3.2. Incubation/early brood rearing period and four week post-hatch poult Kaplan-Meier survival rate estimates (SE) pooled across years for radio-equipped female wild turkeys in east-central Minnesota, USA, 2010-2013.

	Incubation/early Brood		Poult	
	n	2010-2013	*n*	2010-2013
Female (combined)	31	0.59 (0.09)	216	0.26 (0.03)
Adult[a]	11	0.58 (0.14)	106	0.24 (0.04)
Juvenile[a]	20	0.57 (0.11)	110	0.29 (0.04)
Urban	9	0.50 (0.16)	41	0.31 (0.07)
Suburban	14	0.53 (0.12)	95	0.25 (0.04)
Rural	8	0.67 (0.16)	80	0.25 (0.05)

[a] Age-class is defined as Adult (≥1 year of age) or Juvenile (<1 year of age).

Table 3.3. Cause-specific mortality of radio-equipped female wild turkeys in east-central Minnesota, USA, 2010-2013.

Mortality Cause	Females Combined	Adult[a]	Juvenile[a]
Mammalian Predation	16 (53.3%)	8 (61.5%)	8 (47.1%)
Avian Predation	3 (10.0%)	1 (7.7%)	2 (11.8%)
Vehicular Strike	7 (23.3%)	3 (23.0%)	4 (23.5%)
Harvest	1 (3.3%)	1 (7.7%)	0 (0.0%)
Other (unknown)	3 (10.0%)	0 (0.0%)	3 (17.6%)
Total	30 (100%)	13 (43.3%)	17 (56.7%)

[a] Age-class is defined as Adult (≥1 year of age) or Juvenile (<1 year of age).

Table 3.4. Seasonal mortality for radio-equipped wild turkey in east-central Minnesota, USA, for 2010-2013 for spring (1 Apr – 30 Jun), summer (1 Jul – 30 Sep), autumn (1 Oct – 31 Dec) and winter (1 Jan – 31 Mar).

Season	No. Mortalities	Mammalian Predation	Avian Predation	Vehicle Strike	Other
Spring	20	12 (60%)	1 (5%)	5 (25%)	2 (10%)
Summer	5	3 (60%)	1 (20%)	1 (20%)	0 (0%)
Autumn	1	0 (0%)	0 (0%)	0 (0%)	1 (100%)
Winter	4	1 (25%)	1 (25%)	1 (25%)	1 (25%)
Total	30	16 (54%)	3 (10%)	7 (23%)	4 (13%)

Table 3.5. Cause-specific mortality of radio-equipped wild turkey in east-central Minnesota, USA, for 2010-2013. Annual mortality includes annual cause-specific mortality. Incubation/early brood rearing period mortality includes only cause-specific mortality of reproductively active female wild turkeys during the incubation and early brood rearing periods, defined as 28 days pre-hatch and 28 days post-hatch.

Mortality Agent	Annual Mortality			Incubation/early brood rearing Mortality		
	Urban	Suburban	Rural	Urban	Suburban	Rural
Mammalian Predation	6 (85.7%)	4 (36.4%)	6 (50.0%)	4 (100%)	6 (75.0%)	2 (50.0%)

Avian Predation	0 (0.0%)	1 (9.1%)	2 (16.7%)	0 (0.0%)	1 (12.5%)	0 (0.0%)
Vehicular Strike	1 (14.3%)	4 (36.4%)	2 (16.7%)	0 (0.0%)	1 (12.5%)	2 (50.0%)
Harvest	0 (0.0%)	0 (0.0%)	1 (8.3%)	0 (0.0%)	0 (0.0%)	0 (0.0%)
Unknown	0 (0.0%)	2 (18.2%)	1 (8.3%)	0 (0.0%)	0 (0.0%)	0 (0.0%)
Total	7	11	12	4	8	4

Figure 3.1. Incubation and early brood rearing survival distribution for 31 reproductively active female wild turkeys in east-central Minnesota, USA, for 2010-2013. Only female wild turkeys with mammalian cause-specific mortality were included.

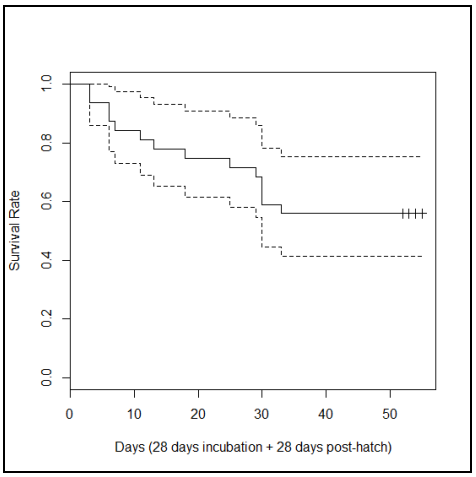
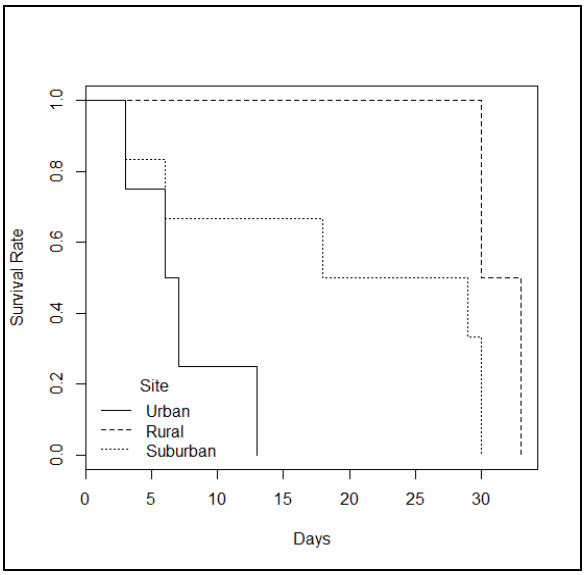


Figure 3.2. Incubation/early brood rearing survival distribution for twelve female wild turkeys with mammalian cause-specific mortality in east-central Minnesota, USA, by study area for 2010 - 2013.



Chapter 4: Range size and habitat use of Wild Turkeys in rural, suburban, and urban landscapes in central Minnesota.

Introduction

Urban wildlife managers are increasingly challenged with managing urban habitats, not only for conservation of biodiversity, but also to improve urban quality of life. Healthy, bio-diverse urban landscapes provide a variety of benefits to local residents, including recreational opportunities, aesthetic landscape value, and connecting urban residents to wildlife. This has generated interest by many public agencies in restoring existing urban habitat to a more 'natural' state and actively planning for inclusion of wildlife preserves as part of future development plans.

As urbanization changes the urban-rural interface into a human-dominated landscape, a greater understanding of urban ecology is required to ensure effective management. Not long ago, we believed urban environments were only suitable for a small subset of species; a presumption no longer held as valid (Ditchkoff et al. 2006). Instead a wide variety of species have entered the urban landscape, including species once thought of as intolerant of human presence (e.g., coyote (*Canis latrans*), white-tailed deer (*Odocoileus virginianus*), bald eagle (*Haliaeetus leucocephalus*)). While many urban residents appreciate wildlife sightings, the ability by some species to exploit the urban environment in such close contact with dense human populations can result in undesirable consequences.

Over the past several decades, researchers have begun to examine how wildlife species in urban areas alter behavioral patterns from their rural counterparts (Leedy 1979, Grinder and Krausman 2001, Chace and Walsh 2004, Ditchkoff et al. 2006). Understanding how species behavior changes in response to urban habitats is an important first step in developing comprehensive wildlife management and conservation decisions specific to the urban setting. Crucial to understanding these responses is knowledge of home range characteristics and habitat use, which are interrelated to the spatial arrangement of resources, resource availability, and local habitat quality (Brown 1980).

The wild turkey (*Meleagris gallopavo*) serves as an excellent example of a rural species now found in many urban areas. Little debate remains regarding economic and recreational benefits wild turkey populations provide in rural settings (Minnesota Department of Natural Resources 2007). However, similar to many of the other non-urban species that have joined the urban setting (Kellert 1985, Conover and Chasko 1985, Clergeau et al. 1998), the entrance of the wild turkey to the Minneapolis-St Paul, Minnesota metropolitan area has been met with mixed emotions. While often a welcome sight in many neighborhoods, negative reactions tend to center on nuisance behaviors, including possible overabundance and perceived property damage (Minnesota Department of Natural Resources 2011).

To date, the vast majority of wild turkey research has been completed in rural and agricultural settings (Little and Varland 1981, Vander Haegen et al. 1988, Thomas and

Litvaitis 1993, Wright et al. 1996, Thogmartin and Schaeffer 2000, Wright and Vangilder 2001, Hubert 2004, Wilson et al. 2005, Humberg et al. 2009), which may not be representative of wild turkey biology in urban landscapes. However, previous research demonstrates that regardless of rural habitat used wild turkeys are dependent upon on seasonal availability of food resources and suitable nesting habitat (Porter et al. 1983, Kurzejeski and Lewis 1990). In addition, the degree of tolerance exhibited by wild turkey populations towards human presence or increased activity remains unclear. Several authors have noted a general wariness of the species to human presence (Mosby and Handley 1943, Wunz 1971, Wright and Speake 1976, Badyaev et al. 1996, Thogmartin 2001), whereas other authors assert a tolerance towards human activity in areas of low hunting pressure (Williams et al. 1971, Dickson et al. 1978). Indeed, in more northern climates some forms of human activity provide benefits to rural wild turkey populations, including access to supplemental food resources from agricultural operations during periods of heavy snow or prolonged winter weather (Wunz and Hayden 1975, Porter et al. 1980, Vander Haegen et al. 1988, Haroldson 1996).

With the increase in wild turkey populations in urban areas, we need to prepare for possible overabundance or nuisance behavior as we increasingly manage both our existing urban lands to support wildlife, and the addition of new areas as urbanization continues to spread. Therefore, my objectives were to assess the relative importance of human-developed habitat to wild turkeys in areas of differing degrees of urbanization by:

1) evaluating the size of home ranges in areas of varying urbanization and 2) assessing habitat use within these home ranges.

Methods

Study Areas

The Lake Elmo rural-fringe (hereafter, "rural") study site is located in Washington County, Minnesota; the Snail Lake (suburban) and Battle Creek (urban) study sites are located in Ramsey County, Minnesota. The rural site serves as a reference site because most of its area consists of agricultural use, large tracts of low-use recreational parkland and natural areas, and low-density residential areas. Parkland in the rural site includes Lake Elmo Park Reserve, a 2,165 acre mixed-use recreational area, and several large areas of maintained grassland and mixed hardwood stands owned by the Minnesota Department of Transportation; Minnesota Correctional Facility, Oak Park Heights, Minnesota; or the city of Bayport, Minnesota.

The suburban site encompasses several county park units, including Snail Lake and Grass Lake Regional Parks, and the Arden Hills Army Training Site (AHATS). This study area is characterized by mostly residential neighborhoods interspersed with parkland of varying recreational or military use. Civilian parkland ranges from high-use recreational areas (mowed lawns and highly maintained vegetation) to low-use parkland managed for native plant species. AHATS is a 1,500 acre site leased by the Minnesota National Guard

for training purposes. Large sections of AHATS are managed for native plant species, including prairie grassland and oak savannah species.

The urban site is centered mainly on Battle Creek Regional Park; however, it includes sections of the National Park Service's Mississippi River and Recreation Area, Minnesota Department of Natural Resources land, and City of St. Paul land. The urban site consists mostly of high-use recreational areas, which are surrounded by dense residential neighborhoods. This site contains small quantities of oak and mixed hardwood woodlands, wetlands, and grassland areas.

Capture and Monitoring

I captured wild turkeys from early December through late March between 2010 and 2012 using air-netting, drop nets, and walk-in live traps (Glazener et al. 1964, Bailey et al. 1980, Gaunt et al. 1999, Nicholson et al. 2000). At capture, I classified each bird by sex and age-class defined as either juvenile (less than one year of age) or adult (after their first nesting season) based on feather characteristics (Williams 1961, Brenneman 1992, Pelham and Dickson 1992, Schroeder and Robb 2005). I fitted all female birds with 78 g motion-sensitive VHF radio transmitters (Advanced Telemetry Systems, Inc., Isanti, Minnesota) using a backpack configuration (Roberts and Porter 1996, Wilson and Norman 1996, Norman et al. 1997). I handled and released all birds at the capture site according to an approved University of Minnesota Animal Research and Care Protocol (IACUC #0911A74374).

I used hand-held receivers and 3-element Yagi antennas to monitor survival and locate radio-tagged females at least three times per week during spring and summer (1 April to 30 September) and at least two times per week during autumn and winter (1 October to 31 March). I located hens using visual observation, signal strength, or triangulation from 3-5 telemetry points within a 20-minute interval to minimize error from movement. I generated location estimates and error polygons using LOAS (Ecological Software Solutions LLC. Hegymagas, Hungary. Version 4.0.3.8), retaining location estimates with an error polygon ≤ 6000 m^2. I estimated an average telemetry error of $\pm$ 3.7° by comparing azimuths of radio-transmitters in known locations for each study area to the true azimuth prior to the onset of the study.

Data Analysis

I did not include any females in the data analysis if fatality occurred within 14 days of capture. I assumed mortality during this period was associated with captured-related stress or transmitter/harness complications (Nenno and Healy 1980, Roberts et al. 1995, Miller et al. 1996). If a monitored individual survived an annual cycle (1 April to 31 March), I then considered activities commencing on 1 April as independent observations from the prior year.

Home Range Analysis

I examined annual and bi-annual (spring/summer and autumn/winter) periods based on biologically relevant periods associated with seasonal activities and shifts in habitat use of wild turkeys on my study areas and previous research (Miller et al. 1997, Miller et al. 1999). I defined the interval occurring between 1 April and 30 September as spring/summer and the interval occurring between 1 October and 31 March as autumn/winter. I generated annual home range estimates based on an annual cycle beginning 1 April and ending 31 March for each female by age-class and study area for the period of 2010-2013. For hens with distinct dispersal or migratory movement, I determined bi-annual periods based on the individual hen's movements. To maintain sample sizes, I combined age class and year for all analyses (White and Garrott 1990).

I calculated annual and bi-annual home range estimates using a convex hull estimator (Worton 1995, Getz and Wilmers 2004). I used a 90% convex hull estimator based on area-observation curves generated for each home range, which ensured that the number of locations I recorded reached an asymptote and therefore sufficient to describe the size of that home range (Odum and Kuenzler 1955). I deemed that for annual home ranges, females with ≥ 30 recorded locations was adequate (e.g., Miller et al. 1997, Seamen et al. 1999, Miller and Conner 2005). For bi-annual home ranges, I deemed females with ≥ 20 recorded locations was adequate (e.g., Hubert 2004, Miller and Conner 2005, Wilson et al. 2005). I used nonparametric approaches for home range analyses because data samples were relatively small and not distributed normally after standard transformation. I compared home range size among study areas using Kruskal-Wallis test methods and *t*-tests for age-class.

Habitat Use Analysis

I created a series of habitat layers for each study area from aerial photographs and the Minnesota Department of Natural Resources' Minnesota Land Cover Classification System (MLCCS) thematic layer (MnDNR Data Deli; http://www.deli.dnr.state.mn.us). I defined each thematic layer by variables grouped into two sets: 1) habitat variables, which represented the percentage of each general habitat type, 2) disturbance variables which represented human activity on the landscape. All measured variables are defined in Table 4.1.

For each season (i.e., annual, spring/summer, and autumn/winter), I determined the number of females monitored and generated an equal number of randomly placed points within their respective study area. I buffered each point to create a circle of equal size as one of the monitored female's home ranges for that study area. While one random circle matched one female's home range, I did not consider these paired observations (Katnit and Wielgus 2005, Miller and Conner 2007).

Using logistic regression (function 'glm' in Project R 2.15.2), I constructed two candidate models (Disturbance and Habitat) that included combinations of habitat and land cover type variables (Table 4.1). I included study area (SITE) as a fixed effect in each candidate model to test for variability among study areas (Hosmer and Lemeshow 2000). I used a theoretic approach to model selection based on Akaike's Information Criterion with a

finite sample size correction (AIC_c) to rank models by degrees of support (Burnham and Anderson 2002). I considered the model with the lowest AIC*c* to be the best supported by the data (Burnham and Anderson 2002). I considered models within two AIC_c points of the top model as competitors for which I computed Akaike weights (w_i) to provide weights of evidence in support of each model (Burnham and Anderson 2002). If the 95% confidence interval of a variable's odds ratio included 1, I deemed that variable to be uninformative (Hosmer and Lemeshow 2000). I built a final model (Global) using the set of all plausible predictors indicated in top and competitive Disturbance and Habitat models. Predictors identified in the final global model set were considered as plausible predictors in relationship to habitat use. Lastly, I used the area under the receiver operating characteristic curve (ROC) to further evaluate the predictive accuracy of all models (Guisan and Zimmermann 2000, Hosmer and Lemeshow 2000, Pearce and Ferrier 2000).

All data analyses were completed in Project R 2.15.2 (R Development Core Team 2012) or ArcGIS 10 (Environmental Systems Research Institute, Redlands, California).

Results

I captured 60 female wild turkeys from 2010-2012. Of the 60 females captured, I used 28 females for the annual home range analysis, 37 for the spring/summer home range analysis, and 28 autumn/winter home range analysis.

Annual and Bi-seasonal Home Range

Annual Home Ranges

Across all study areas and all female wild turkeys, I observed an average annual home range size of 41.3 hectares (ha; $n = 28$). I found annual home range size for suburban females (64.5 ha, $n = 9$) was larger than rural (38.0 ha, $n = 11$) or urban females (19.6 ha, $n = 8$), with home range size differing between study areas ($\chi^2 = 12.26$, $DF = 2$, $P = 0.002$) (Table 4.2). A pairwise comparison of study areas determined that annual home range size differed significantly between suburban females and urban females ($P = 0.002$), but not between urban females and rural females ($P = 0.23$) or suburban females and rural females ($P = 0.13$). I found annual home range size for adult females (29.9 ha, $n = 9$) was larger than juvenile females (28.3 ha, $n = 19$), but did not differ significantly ($W = 94$, $P = 0.70$). I found annual home range size for females that attempted to nest (41.2 ha, $n = 18$) was larger than females that did not attempt to nest (18.4 ha, $n = 10$), and differed significantly ($W = 27$, $P = 0.002$).

Bi-annual Home Ranges

Spring/summer home ranges included both females that attempted to nest, brooding hens, and non-reproductively active females. Across all study areas and all female wild turkeys, I observed an average spring/summer home range size of 26.4 ha ($n =37$) (Table 4.2). I found spring/summer home range size for suburban females (44.8 ha, $n = 11$) was larger than rural (23.0 ha, $n = 17$) or urban females (10.3 ha, $n = 9$), although home range size

did not differ significantly (χ^2 = 5.98, *DF* = 2, *P* = 0.056). I found spring/summer home range size for adult females (30.6 ha, *n* = 15) was larger than juvenile females (23.6 ha, *n* = 22), but did not differ significantly (*W* = 175, *P* = 0.77). I found spring/summer home range size for females that attempted to nest (29.9 ha, *n* = 27) was larger than females that did not attempt to nest (16.8 ha, *n* = 10), but did not differ significantly (*W* = 133, *P* = 0.96).

Across all study areas and all female wild turkeys, I observed an average autumn/winter home range size of 25.1 ha (*n* = 28) (Table 4.2). I found autumn/winter home range size for suburban females (30.9 ha, *n* = 9) was larger than rural (28.6 ha, *n* = 11) or urban females (13.8 ha, *n* = 8), with home range size not differing between study areas (χ^2 = 2.71, *DF* = 2, *P* = 0.26). I found autumn/winter home range size for adult females (24.6 ha, *n* = 10) was larger than juvenile females (12.7 ha, *n* = 18), but did not differ significantly (*W* = 117, *P* = 0.13).

Annual and Bi-annual Habitat Use

Annual Habitat Use

For annual habitat use based on disturbance, I deemed two models (Annual Disturbance I and Annual Disturbance II) competitive (AICc ≤ 2 points) with the top model (Annual Disturbance III) (Table 4.3). I considered all variables contained in the top and competitive annual disturbance models (low-use park, low-density residential, high-density residential, and agricultural) as plausible predictors of habitat use. The confidence interval (CI) for the estimated odds ratio for the agricultural variable in Annual

Disturbance II model included 1, indicating this variable may not be well supported (Table 4.4). The CI for the estimated odds ratio for high-density residential in one of two models (Annual Disturbance III) it was selected for included 1, indicating this variable may not be well supported (Table 4.4). I found the ROC scores produced for Annual Disturbance I ([Park Low-use + Residential Low-density]; ROC = 0.77), Annual Disturbance II ([Park Low-use + Residential Low-density + Residential High-density + Agricultural]; ROC = 0.74), and Annual Disturbance III ([Park Low-use + Residential Low-density + Residential High-density]; ROC = 0.77) did not differ significantly (P = 0.61, P = 0.63, P = 0.99; replications = 10,000), which suggests Agricultural and Residential High-density may not be important in discriminating the likelihood of annual habitat use. I found the statistically supported variables in the top annual disturbance model were low-use park (P = 0.002), low-density residential (P = 0.010), and high-density residential. I found habitat use was positively related to low-use park, low-density residential, and high-density residential. Inclusion of the SITE variable did not produce significant results for either the full or top models.

For annual habitat use based on habitat variables, I deemed two models (Annual Habitat I and Annual Habitat II) competitive (AICc $\leq$ 2 points) with the top model (Annual Habitat III) (Table 4.3). I considered all variables contained in the top and competitive annual habitat models (deciduous tree, conifer tree, maintained grass, non-maintained grass, and residential) as plausible predictors of habitat use. The CI for the estimated odds ratio for maintained grass (Annual Habitat II) and non-maintained grass (Annual Habitat I)

variables included 1, indicating these variables may not be well supported (Table 4.4). I found the ROC scores produced for Annual Habitat I ([Tree, Deciduous + Tree, Conifer + Residential + Grass Non-maintained]; ROC = 0.70), Annual Habitat II ([Tree, Deciduous + Tree, Conifer + Residential + Grass Maintained]; ROC = 0.64), and Annual Habitat III ([Tree, Deciduous + Tree, Conifer + Residential]; ROC = 0.71) did not differ significantly ($P = 0.40$, $P = 0.96$, $P = 0.06$; replications = 10,000), which suggests Grass Non-maintained and Grass Maintained may not be important in discriminating the likelihood of annual habitat use. I found the statistically supported variables in the top annual habitat model were deciduous tree ($P = 0.009$), conifer tree ($P = 0.015$), and residential ($P = 0.034$). I found habitat use was positively related to deciduous tree, conifer tree, and residential. Inclusion of the SITE variable did not produce significant results for either the full or top model.

I created an annual global model including all plausible predictive variables from the top and competitive annual disturbance and annual habitat models. I determined a single model (Annual Global I) competitive (AICc $\leq$2 points) with the top model (Annual Global II) (Table 4.3). I considered all variables contained in the top and competitive annual global models (conifer tree, non-maintained grass, residential, high density residential, and low-use park) as plausible predictors of habitat use. I found the ROC scores produced for Annual Global I ([Residential + Residential High-density + Park Low-use + Grass Non-maintained]; ROC = 0.63) and Annual Global II ([Residential + Residential High-density + Park Low-use + Grass Maintained + Tree, Conifer]; ROC =

0.66) did not differ significantly ($P = 0.12$; replications = 10,000), which suggests Tree, Conifer, may not be important in discriminating the likelihood of annual habitat use. I found the statistically supported variables in the top global model were residential ($P = 0.014$), low-use park (P = 0.001), and non-maintained grass ($P = 0.020$) (Table 4.4). I found habitat use was positively related to residential and low-use park, and negatively related to non-maintained grass. Inclusion of the SITE variable did not produce significant results for either the full or top model.

Spring/Summer Habitat Use

For spring/summer habitat use based on disturbance, I determined a single model (Spring/Summer Disturbance I) as best (Table 4.3). I found the most predictive variables in the top model were low-use park, and low-density residential. The CI for the estimated odds ratio for the low-density residential variable in the top model included 1, indicating this variable may not be well supported (Table 4.4). I found the statistically supported variables in the top spring/summer disturbance model was low-use park ($P = 0.001$) (Table 4.4). I found habitat use was positively related to low-use park. Inclusion of the SITE variable did not produce significant results for either the full or top models.

For spring/summer habitat use based on habitat variables, I deemed two models (Spring/Summer Habitat I and Spring/Summer Habitat II) competitive (AICc $\leq$ 2 points) with the top model (Spring/Summer Habitat III) (Table 4.3). I considered all variables contained in the top and competitive spring/summer habitat models (deciduous tree, conifer tree, non-maintained grass, and residential) as plausible predictors of

spring/summer habitat use. The CI for the estimated odds ratio for the residential variable in Spring/Summer Habitat II and grass non-maintained variable in Spring/Summer Habitat III included 1, indicating these variables may not be well supported (Table 4.4). I found the ROC scores produced for Spring/Summer Habitat I ([Tree, Deciduous + Tree, Conifer]; ROC = 0.69), Spring/Summer Habitat II ([Tree, Deciduous + Tree, Conifer + Residential + Grass Non-maintained]; ROC = 0.59), and Spring/Summer Habitat III ([Tree, Deciduous + Tree, Conifer + Grass Non-maintained]; ROC = 0.69) did not differ significantly (P = 0.22, 0.18, 0.99; replications = 10,000), which suggests Grass Non-maintained and Residential may not be important in discriminating the likelihood of spring/summer habitat use. I found the statistically supported variables in the top and competitive spring/summer habitat models were deciduous tree (P = 0.017), conifer tree (P = 0.028), and non-maintained grass (P = 0.026). I found habitat use was positively related to deciduous tree, conifer tree, and non-maintained grass. Inclusion of the SITE variable did not produce significant results for either the full or top model.

I created a global spring/summer model including all variables from the top and competitive spring/summer disturbance and habitat global models, and determined a single top global model (Spring/Summer Global I) (Table 4.3). I considered all variables contained in the top spring/summer global model (deciduous tree, conifer tree, low-density residential, and low-use park) as plausible predictors of habitat use. The CI for the estimated odds ratio for the deciduous tree and low-density residential variables in top model included 1, indicating these variables may not be well supported (Table 4.4). I

found the statistically supported variables in the top spring/summer global model were conifer tree (P = 0.017), residential low-density (P = 0.018) and low-use park (P = 0.002). I found habitat use was positively related to conifer tree, and low-use park. Inclusion of the SITE variable did not produce significant results for either the full or top model.

Autumn/Winter Habitat Use

For autumn/winter habitat use based on disturbance, I determined a single model (Autumn/Winter Disturbance I) competitive (AICc ≤ 2 points) with the top model (Autumn/Winter Disturbance II) (Table 4.3). I considered all variables contained in the top and competitive autumn/winter disturbance models (low-use park, low-density residential, high-density residential) as plausible predictors of habitat use. The CI for the estimated odds ratio for the high-density residential variable in the top autumn/winter disturbance model included 1, indicating this variable may not be well supported (Table 4.4). I found the ROC scores produced for Autumn/Winter Disturbance I ([Park Low-use + Residential Low-density]; ROC = 0.74) and Autumn/Winter Disturbance II ([Park Low-use + Residential Low-density]; ROC = 0.74) did not differ significantly (P = 0.99; replications = 10,000), which suggests Residential High-density may not be important in discriminating the likelihood of autumn and winter habitat use. I found the statistically supported variables were low-density residential (P = 0.002) and low-use park (P = 0.003). I found habitat use was positively related to low-density residential and low-use

park. Inclusion of the SITE variable did not produce significant results for either the full or top models.

For autumn/winter habitat use based on habitat variables, I determined a single top model (Autumn/Winter Habitat I) as best (Table 4.3). I considered all variables contained in the top autumn/winter habitat model (deciduous tree, conifer tree, residential, livestock, grass non-maintained, grass maintained, and water) as plausible predictors of autumn/winter habitat use. The CI for the estimated odds ratio for the water variable in the top autumn/winter habitat model included 1, indicating this variable may not be well supported (Table 4.4). I found the most predictive variables in the top habitat specialist model were deciduous tree ($P = 0.019$), conifer tree ($P = 0.015$), residential ($P = 0.017$), livestock ($P = 0.043$), non-maintained grass ($P = 0.026$), and maintained grass ($P = 0.031$) (Table 4.4). I found habitat use was positively related to deciduous tree, conifer tree, residential, livestock, non-maintained grass, and maintained grass. Inclusion of the SITE variable did not produce significant results for either the full or top model.

I created a global model including all top and competitive autumn/winter model variables, and determined a single model (Autumn/Winter Global I) as best (Table 4.3). The most predictive variables in the Autumn/Winter Global I model were low-density residential, high-density residential, low-use park, and livestock. The variables with statistically significant support were low-density residential ($P = 0.003$), high-density residential ($P = 0.020$), low-use park ($P = 0.003$), and livestock ($P = 0.037$) (Table 4.4). I

found habitat use was positively related to low-density residential, high-density residential, low-use park, and livestock. Inclusion of the SITE variable did not produce significant results for either the full or top model.

Discussion

As urbanization continues to impact a larger proportion of the planet (Brown et al. 2005), there is a growing need to understand how wildlife responds to this novel landscape. In the past several decades, research has increasingly focused on the effects of urbanization on home range size and habitat use. However, the influence of urban land change on home range size remains unclear. For example, numerous studies suggest home range size in urban areas are typically smaller than in rural areas (i.e., white-tail deer [Loft et al. 1984, Cornecelli 1992], coyote [Atwood 2004, Gehrt et al. 2009], raccoons [*Procyon lotor*; Gehrt 2004]), conversely other studies have reported inconclusive or no change (coyote [Grinder and Krausman 2001], striped skunk [*Mephitis mephitis*; Gehrt 2004]).

My home range estimates should be interpreted with caution, as the inherent variability and choice of home range estimators, sample size, and relevant biological or environmental variables makes direct comparisons among other wild turkey studies somewhat speculative in nature. However, previous wild turkey research indicates a strong trend for smaller home ranges in higher quality habitat or in areas with diverse habitats (Barwick and Speake 1973, Speake et al. 1975, Everett et al. 1979, Brown 1980, Miller and Conner 2005).

Similar to previous wild turkey research in rural landscapes (Barwick and Speake 1973, Brown 1980, Miller et al. 1997, Hubert 2004), I observed a high degree of variation in seasonal and annual home ranges estimates. On average, I found annual home ranges for this study were smaller than reported in previous research (Barwick and Speake 1973, Miller et al. 1997, Paisley et al. 2000, Miller and Conner 2005, Wilson et al. 2005). Comparisons among other wildlife species suggests a general trend for smaller home ranges to occur on urban landscapes (Riley et al. 2003, Gehrt et al. 2009). Similarly, I found monitored urban females tended to maintain a smaller home range than suburban or rural females. My findings, however, contradict the contention that smaller home ranges occur on more developed landscapes, as suburban females typically maintained a much larger home range than rural females.

In general, home range size is a reflection of the spatial location of resources, distance between resources, habitat fragmentation, and often avoidance of human activities. Urbanization is directly associated with these processes. I hypothesize that the observed reduction in urban wild turkey home range size was attributable to avoidance of direct human interaction and the concentration of suitable habitat and available resources into smaller more ‘natural’ park-like habitat patches. While urban females did exhibit a greater tolerance of human presence than rural birds, they still remained much more wary of human activities than suburban females. For example, I rarely observed urban females

entering residential areas more than a few hundred meters from an edge of a park unit at any point during the year.

This contrasted sharply with suburban females, which routinely resided in residential areas for extended periods, with some females using residential areas for their full life-cycle. The behavior of suburban females to maintain larger home ranges also contrasts to other species that have colonized the urban landscape (i.e., white-tail deer [Loft et al. 1984, Cornecelli 1992], raccoons [Gehrt 2004], where their home range size is smaller than commonly reported for rural areas. Although Grinder and Krausman (2001) found urban coyote home ranges in Tucson, Arizona, were on average comparable in size as coyotes in other habitat types.

I also found reproductively active females annual home range use differed from non-reproductively active females, with reproductively active females using a larger area (41.2 ha vs 18.4 ha). Although not statistical different, I found a similar trend for spring/summer home range estimates (29.9 ha vs 16.8 ha). The larger average home range size of reproductively active females compared to non-reproductively active females is likely attributable to reproductively active females seeking suitable nesting (Cobb and Doerr 1997) and brood rearing habitat (Miller et al. 1997). For example, Badyaev et al (1996) found female wild turkeys that sampled more habitats, as demonstrated in part by having a greater degree of area covered prior to nesting, acquired higher-quality nesting habitat. My results suggest variation in annual and spring/summer

home range size between reproductively and non-reproductively active females is due in part to the complex interaction of individual behavior related to habitat sampling and spatial location of suitable resources.

Habitat use

Urbanization profoundly alters habitats, with urban habitats differing markedly from rural or more natural habitats (Marzluff and Ewing 2001, Shochat et al. 2006). Wild turkey populations occur in a diverse set of rural habitats; however, range use must include suitable habitat to fulfill females life-cycle needs. I observed an apparent shift in habitat use between spring/summer and autumn/winter with a greater degree of more 'natural-like' habitat (i.e., parkland, conifer tree) predictive of spring/summer use and developed habitat (i.e., residential areas, agricultural) predictive of autumn/winter use. Wild turkey range shifts in rural environments have been linked to resource availability (Porter 1977), particularly nesting and brood habitats (Smith et al. 1990). The observed range shift supports the general contention that wild turkeys are more tolerant of highly developed, human dominated areas than once believed (Mosby and Handley 1943). Across all study areas, monitored wild turkeys used residential and agricultural areas, when available. However, similar to other authors observations (Wunz 1971, Wright and Speake 1976) all flocks observed (includes radio and non-radio tagged females) appeared to constrict their home ranges or abandon established home ranges in areas of increased human use. For example, during sewer replacement in a remote area of Battle Creek Park, wild turkey constricted their normal movements to areas north and south of the work area, largely

abandoning the corridor that was used to a high degree prior to commencement of the sewer maintenance. Within months of work completion, this flock once again reclaimed this portion of their home range.

Wild turkey habitat requirements are most specific during the reproductive period. My spring/summer global model selection results show a discriminatory use of more 'natural-like' habitat (i.e., parkland, conifer tree) during the spring/summer period. Specifically, model inference indicated low-use parkland and presence of conifer trees were positively related to habitat use. The shift to low-use park areas for nesting is consistent with much of the published wild turkey literature, which indicates females are likely to abandon nesting attempts if flushed or disturbed. I found this behavior consistent with my rural females where one monitored female abandoned her nest after a single flushing incident. Conversely, I noted a much greater degree of tolerance with several of my urban and suburban female wild turkeys, with one nesting female tolerant of a resident repeatedly approaching and photographing her, often approaching to within a meter during the entire incubation cycle.

I found a shift in autumn/winter habitat use include more developed areas, including residential and agricultural areas. Presumably, overwintering habitat use is a complex relationship between available or accessible food resources, available night roosting locations near food resources, and the influence of adverse weather conditions (i.e., snow pack depth, temperature). Porter (1977) found weather conditions restricted wild turkey

movements and home range size in central Minnesota. I observed a similar pattern with females restricting their movements during harsh winter weather to areas near reliable food resources.

One food resource available on the urban landscape in greater density than rural areas is the presence of bird feeders. While I observed wild turkeys foraging on a diverse set of food resources in urban areas (i.e., acorns, bird feeders, crab apple trees, apple trees, sumac) during the winter months, I consistently observed the birds foraging near bird feeders or other human provided food plots. Reliance upon bird feeders and other human provided food resources was evident once snow depth was greater than 20 cm, with suburban and urban females abandoning park use to a large degree, usually only using the fringe areas as corridors. Importantly, reliance on human provided food resources concentrated the birds into relatively small areas on the residential landscape. This massing of birds into small ranges likely had the undesirable effect of creating human-wild turkey conflicts. As one neighborhood residence stated "a few wild turkeys in the yard is an enjoyable experience, however 30, 40, or 50 wild turkeys at a time is just a nuisance."

Conclusions

With the rapid expansion of urban areas, there is a greater interest in preserving and developing more natural-like areas within our cities. Indeed, we have observed a rise in species once thought intolerant of human activities successfully enter the urban

landscape. Importantly, to effectively manage, or enhance urban habitat for wildlife inclusion into the urban landscape requires a basic understanding of how species respond and potentially benefit from this novel habitat. This study reaffirms the truly adaptive nature of the wild turkey, and its ability to exploit a landscape dominated by human development and activity.

My results can be summarized into two primary observations. First, in a landscape dominated by human development, natural habitat was used heavily by female wild turkeys. Second, despite the importance of natural habitat for urban female wild turkeys, most individuals exhibited a high reliance on developed, human-dominated areas. The true tolerance of wild turkeys to human presences remains unclear. While females consistently demonstrated a reliance on developed areas during autumn and winter, I interpret the shift in habitat use to low-use parkland as support of avoidance of areas associated with human activity.

Urban wildlife managers should consider these factors when managing urban wild turkey populations, or in planning enhancements to urban park systems. Planners need consider seasonal movements of species to fulfill basic life-cycle needs, which is likely to be of greater importance on a highly fragmented urban landscape. In addition, the close proximity of residential areas are likely to cause greater conflict with human populations, especially for species successful in exploiting urban resources.

From my results and personal observations, it is clear that urban wild turkeys were using a variety of strategies to exploit the urban landscape, with some populations becoming large enough to cause nuisance behavior. The high reliance on human dominated areas during autumn and winter for supplemental dietary resources provides an excellent management tool for control of nuisance populations. I suggest that public wildlife management agencies should consider a public outreach program in areas with nuisance turkeys. Future research should further explore the limitations of wild turkeys' use of urban landscapes, including examination of food and habitat resources during all seasonal periods.

Tables and Figures

Table 4.1. Measured habitat variables grouped into two sets for evaluating the variables influence on habitat use by Wild Turkeys in central Minnesota, 2010- 2013.

Variable Name	Description
Disturbance/Human Presence Candidate Model Variable Names	
Park, Low-use	Areas not actively maintained, areas not within 50 m of a recreational trail, or trail areas used ≤ 1 person per hour
Park, High-use	Actively maintained areas, recreational areas, areas within 50 m of a trail used at a rate of > 1 person per hour
Residential, Low-density	Average house lot size ≤ 0.5 ac per residential unit
Residential, High-density	Average residential lot size > 0.5 ac per residential unit
Agricultural	Land use dominated by row crops or livestock operations
SITE	Study area indicator
Habitat Candidate Model Variable Names	
Residential	Land cover dominated by human use (i.e., housing, light industry, retail)
Tree, deciduous	Land cover dominated by deciduous tree species
Tree, conifer	Land cover dominated by conifer tree species
Livestock	Land use dominated by livestock operations (i.e., dairy cows, horse

	farms, cattle operations)
Row crop	Land use dominated by row crop production
Grass, Maintained	Maintained grassy areas (i.e., recreational areas, picnic areas, road medians)
Grass, Non-Maintained	Grassy areas not maintained (i.e., planted prairie, road medians, remote park areas)
Water	Open water
SITE	Study area indicator

Table 4.2. Number of female wild turkey, mean number of telemetry locations used to calculate home ranges, mean, minimum and maximum sizes of annual and seasonal home ranges in hectare the Minneapolis-St. Paul, Minnesota, metropolitan area.

Study Area	Year	Season	# Females	Ave # Locations	Mean (SE)	Min.	Max.
Urban	2010	Annual	1	109	9.7	--	--
Urban	2011	Annual	4	78	15.8 (8.7)	8.5	26.6
Urban	2012	Annual	3	73	27.9 (1.8)	26.5	29.9
Urban	2010-2012	Annual	8	80	19.6 (9.2)	29.9	8.5
Urban	2010	Spring/Summer	1	48	9.8	--	--

Study Area	Year	Season	# Females	Ave # Locations	Mean (SE)	Min.	Max.
Urban	2011	Spring/Summer	4	39	11.1 (7.4)	4.8	20.6
Urban	2012	Spring/Summer	4	38	9.5 (6.6)	3.9	18.9
Urban	2010-2012	Spring/Summer	9	40	10.3 (6.1)	3.9	20.6
Urban	2010	Autumn/Winter	1	73	6.8	--	--
Urban	2011	Autumn/Winter	4	39	9.8 (2.7)	6.6	13.0
Urban	2012	Autumn/Winter	3	31	21.3 (8.4)	11.8	27.6
Urban	2010-2012	Autumn/Winter	8	40	13.8 (8.0)	6.6	27.6
Suburban	2010	Annual	1	88	50.0	--	--
Suburban	2011	Annual	2	75	39.8 (4.2)	36.9	42.8
Suburban	2012	Annual	6	88	75.2 (50.1)	28.3	161.8
Suburban	2010-2012	Annual	9	85	64.5 (42.8)	28.3	161.8
Suburban	2010	Spring/Summer	1	35	44.4	--	--
Suburban	2011	Spring/Summer	3	31	9.7 (6.9)	5.8	17.9

Study Area	Year	Season	# Females	Ave # Locations	Mean (SE)	Min.	Max.
Suburban	2012	Spring/Summer	7	43	59.8 (76.2)	8.1	215.2
Suburban	2010-2012	Spring/Summer	11	39	44.8 (63.4)	5.8	215.2
Suburban	2010	Autumn/Winter	1	50	3.8	--	--
Suburban	2011	Autumn/Winter	2	36	9.0 (4.6)	5.7	12.2
Suburban	2012	Autumn/Winter	6	40	42.7 (39.5)	5.9	113.9
Suburban	2010-2012	Autumn/Winter	9	40	30.9 (35.9)	3.8	113.9
Rural	2010	Annual	1	111	54.6	--	--
Rural	2011	Annual	5	73	20.6 (7.3)	12.1	39.6
Rural	2012	Annual	5	79	56.7 (28.6)	25.8	99.0
Rural	2010-2012	Annual	11	79	38.1 (26.0)	12.1	99.0
Rural	2010	Spring/Summer	1	50	14.4	--	--
Rural	2011	Spring/Summer	9	33	15.9	12.7	28.7

Study Area	Year	Season	# Females	Ave # Locations	Mean (SE)	Min.	Max.
					(7.8)		
Rural	2012	Spring/Summer	7	34	33.4 (27.5)	7.5	89.7
Rural	2010-2012	Spring/Summer	17	35	23.0 (19.8)	7.5	89.7
Rural	2010	Autumn/Winter	1	58	17.8	--	--
Rural	2011	Autumn/Winter	5	38	12.5 (5.1)	5.1	18.8
Rural	2012	Autumn/Winter	5	35	46.8 (16.9)	29.8	73.8
Rural	2010-2012	Autumn/Winter	11	38	28.6 (20.8)	5.1	73.8

Table 4.3. Support for models predicting habitat use recorded for 28 Wild Turkeys on Ramsey and Washington counties, Minnesota during 2010-2013. Models are based on Akaike's information criterion corrected for small sample sizes (AIC*c*). I compared the different models with the null-hypothesis model that contained all habitat variables during three temporal periods (annual, spring/summer, autumn/winter). A final hybrid variable set that included habitat variables from all top models evaluated their influence on nest success. *K* is the number of parameters in the model; ΔAIC*c* is the difference in

AIC*c* between each model and the top model; Akaike weight (*wi*) is the weight of the evidence for model *i*.

Model	*K*	AIC_c	ΔAIC_c	w_i	ROC
ANNUAL HABITAT USE MODELS					
Annual Disturbance Presence Full and Top Modes					
Annual Disturbance Full [Park Low-use + Park High-use + Residential Low-density + Residential High-density + Agricultural]	5	65.67	0.00	0.07	0.71[a]
Annual Disturbance I [Park Low-use + Residential Low-density]	2	64.10	1.57	0.14	0.77[b]
Annual Disturbance II [Park Low-use + Residential High-density + Residential Low-density + Agricultural]	4	63.16	2.51	0.24	0.74
Annual Disturbance III [Park Low-use + Residential High-density + Residential Low-density]	3	62.75	2.92	0.29	0.77
Annual Habitat Full and Top Models					
Annual Habitat Full [Water + Tree, Deciduous + Tree, Conifer + Residential + Livestock + Row Crop + Grass Non-maintained + Grass Maintained]	8	74.60	0.00	0.01	0.67[a]
Annual Habitat I [Tree, Deciduous + Tree,	4	66.91	7.69	0.24	0.70[b]

Model	*K*	AIC_c	ΔAIC_c	w_i	ROC
Conifer + Residential + Grass Non-maintained]					
Annual Habitat II [Tree, Deciduous + Tree, Conifer + Residential + Grass Maintained]	4	66.59	8.01	0.28	0.64
Annual Habitat III [Tree, Deciduous + Tree, Conifer + Residential]	3	66.56	8.04	0.29	0.71
Annual Global Full and Top Models					
Global Full [Tree, Deciduous + Tree, Conifer + Residential + Residential Low-density + Residential High-density + Park Low-use + Agricultural + Grass Maintained + Grass Non-maintained]	9	61.95	0.00	0.01	0.78[a]
Global I [Residential + Residential High-density+ Park Low-use + Grass Non-maintained]	4	54.27	7.68	0.18	0.63[b]
Global II [Residential + Residential High-density+ Park Low-use + Grass Non-maintained + Tree, Conifer]	5	52.28	9.67	0.49	0.66
SPRING/SUMMER HABITAT USE MODELS					
Disturbance Full and Top Models					
Spring/Summer Disturbance Full [Park	5	110.62	0.00	0.31	0.65[a]

Model	*K*	AIC_c	ΔAIC_c	w_i	ROC
Low-use + Park High-use + Residential Low-density + Residential High-density+ Agricultural]					
Spring/Summer Disturbance I [Park Low-use + Residential Low-density]	2	109.60	1.02	0.41	0.69
Spring/Summer Habitat Full and Top Models					
Spring/Summer Habitat Full [Water + Tree, Deciduous + Tree, Conifer + Residential + Livestock + Row Crop + Grass Non-maintained + Grass Maintained]	8	118.93	0.00	0.01	0.49[a]
Spring/Summer Habitat I [Tree, Deciduous + Tree, Conifer]	2	113.06	5.87	0.12	0.69[b]
Spring/Summer Habitat II [Tree, Deciduous + Tree, Conifer + Residential + Grass Non-maintained]	4	111.65	7.28	0.24	0.59
Spring/Summer Habitat III [Tree, Deciduous + Tree, Conifer + Grass Non-maintained]	3	111.21	7.72	0.29	0.69
Spring/Summer Global Full and Top Models					
Global Full [Tree, Deciduous + Tree, Conifer + Grass Non-maintained + Park	6	111.17	0.00	0.07	0.65[a]

Model	*K*	AIC_c	ΔAIC_c	w_i	ROC
Low-use + Residential Low-density + Residential]					
Global I [Tree, Deciduous + Tree, Conifer + Park Low-use + Residential Low-density]	4	108.17	3.00	0.73	0.74
AUTUMN/WINTER HABITAT USE MODELS					
Autumn/Winter Disturbance Full and Top Models					
Autumn/Winter Disturbance Full [Park Low-use + Park High-use + Residential Low-density + Residential High-density + Agricultural]	5	66.29	0.00	0.12	0.73[a]
Autumn/Winter Disturbance I [Park Low-use + Residential Low-density]	2	64.61	1.68	0.28	0.74[b]
Autumn/Winter Disturbance II [Park Low-use + Residential Low-density + Residential High-density]	3	63.80	2.49	0.42	0.74
Autumn/Winter Habitat Full and Top Models					
Autumn/Winter Habitat Full [Water + Tree, Deciduous + Tree, Conifer + Residential + Livestock + Row Crop + Grass Non-maintained + Grass Maintained]	8	68.92	0.00	0.38	0.70[a]
Autumn/Winter Habitat I [Water + Tree,	7	67.91	1.01	0.62	0.77

Model	*K*	AIC_c	ΔAIC_c	w_i	ROC
Deciduous + Tree, Conifer + Residential + Livestock + Grass Non-maintained + Grass Maintained]					
Autumn/Winter Global Full and Top Models					
Autumn/Winter Global Full [Tree, Deciduous + Tree, Conifer + Park Low-use + Residential + Residential Low-density + Residential High-density + Grass Non-maintained + Grass Maintained + Livestock + Water]	10	68.21	0.00	0.01	0.85[a]
Autumn/Winter Global I [Park Low-use + Residential Low-density + Residential High-density + Livestock]	4	58.41	9.80	0.99	0.81

[a]*ROC* curves for the full and top competitive model do not differ at a 0.05 level using Delong and bootstrap method (boot *n* = 10000)

[b]*ROC* curves among top competitive models do not differ at a 0.05 level using Delong and bootstrap method (boot *n* = 10000)

Table 4.4. Parameter and odds ratio estimates, including 95% confidence intervals (CI), for habitat variables included in top model(s) of 28 Wild Turkey on Ramsey and Washington counties, Minnesota during the breeding seasons 2010-2013.

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
ANNUAL HABITAT USE MODELS					
Annual Disturbance Top Models					
Annual Disturbance I					
Intercept	-2.62	-4.64, -1.04	0.004	0.07	0.01, 0.35
Park Low-use	4.39	1.73, 7.65	0.003	81.04	5.65, 2.09e+03
Residential Low-density	5.81	1.16, 11.33	0.022	333.45	3.19, 8.36e+04
Annual Disturbance II					
Intercept	-6.27	-11.39, -2.61	0.005	1.13e-05	0.00001, 0.007
Park Low-use	8.47	3.83, 14.69	0.002	4769.97	45.94, 2.41e+06
Residential Low-density	8.85	3.06, 16.22	0.007	6952.24	21.41, 1.11e+07
Residential High-density	6.49	1.11, 13.23	0.031	657.78	3.03, 5.56e+05
Agriculture	4.31	-1.64, 10.73	0.159	74.36	0.19, 4.55e+04

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Annual Disturbance III					
Intercept	-4.59	-8.17, -1.92	0.003	1.01e-02	2.82e-4, 0.15
Park Low-use	6.77	3.00, 11.60	0.002	8.72e+02	20.11, 1.09e+05
Residential Low-density	7.72	2.40, 14.31	0.010	2.25e+03	11.06, 1.64e+06
Residential High-density	4.07	-0.07, 8.77	0.064	58.61	0.93, 6.43e+03
Annual Habitat Top Models					
Annual Habitat I					
Intercept	-5.48	-10.60, -1.63	0.015	4.16	2.49, 1.96
Tree, Deciduous	7.34	2.58, 13.33	0.007	1.54e+03	1.17, 6.16e+05
Tree, Conifer	21.82	7.70, 41.56	0.010	3.01e+09	2.21e+03, 1.12e+18
Residential	6.53	1.44, 13.12	0.026	6.89e+02	4.24, 4.98e+05
Grass, Non-	4.46	-1.50,	0.166	86.51	0.22,

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
maintained		11.43			9.22e+04
Annual Habitat II					
Intercept	-2.31	-4.73, -0.32	0.037	0.01	0.01, 0.73
Tree, Deciduous	4.32	0.73, 8.65	0.029	75.38	2.08, 5.74e+03
Tree, Conifer	16.58	4.51, 33.77	0.022	1.58e+07	90.78, 4.63e+14
Residential	3.27	0.18, 6.89	0.051	26.31	1.20, 987.20
Grass, Maintained	-7.77	-20.07, 1.85	0.154	4.24e-04	1.93e-09, 6.38
Annual Habitat III					
Intercept	-2.98	-5.32, -1.09	0.005	5.06	4.87e-03, 0.34
Tree, Deciduous	5.06	1.55, 9.34	0.009	158.04	4.72, 1.14e+04
Tree, Conifer	16.41	5.06, 32.17	0.015	1.34e+07	157.54, 9.36e+13
Residential	3.55	0.46, 7.15	0.034	34.95	1.58, 1.28e+03

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Annual Global Top Models					
Annual Global I					
Intercept	-7.62	-13.70, -3.31	0.003	4.93e-04	1.12e-06, 0.04
Tree, Conifer	18.19	1.17, 44.94	0.093	7.96e+07	3.23, 3.29e+19
Residential High-density	-9.70	-20.77, -130	0.042	6.10e-05	9.55e-10, 0.27
Residential	16.57	6.34, 31.68	0.008	1.59e+07	569.38, 5.73e+13
Park Low-use	11.57	5.87, 19.39	0.001	1.06e+05	355.33, 2.64e+08
Grass, Non-maintained	-6.11	-13.49, -0.06	0.067	2.22e-03	1.38e-06, 0.94
Annual Global I					
Intercept	-5.57	-10.15, -2.20	0.005	3.81e-03	3.90e-05, 0.11
Residential High-density	-8.32	-18.60, -0.13	0.068	2.43e-04	8.35e-09, 0.87
Residential	12.69	3.92,	0.014	3.25e+05	50.61,

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
		24.92			6.63e+10
Park Low-use	10.99	5.62, 18.41	0.001	5.90e+04	274.56, 9.91e+07
Grass, Non-maintained	-7.90	-15.44, -1.85	0.020	3.71e-04	1.97e-07, 0.16
SPRING/SUMMER HABITAT USE MODELS					
Spring/Summer Disturbance Top Model					
Spring/Summer Disturbance I					
Intercept	-1.75	-3.03, -0.66	0.003	0.17	0.05, 0.52
Park Low-use	3.19	1.48, 5.17	0.001	24.31	4.37, 176.56
Residential Low-density	2.41	-0.21, 5.20	0.076	11.14	0.81, 181.75
Spring/Summer Habitat Top Models					
Spring/Summer Habitat I					
Intercept	-0.79	-1.54, -0.01		0.45	0.21, 0.91
Tree, Deciduous	2.13	0.15, 4.28		8.38	1.16, 71.92
Tree, Conifer	11.80	3.24,		1.33e+05	25.61,

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
		22.82			8.13e+09
Spring/Summer Habitat II					
Intercept	-2.82	-5.43, -0.71	0.017	0.06	0.001, 0.49
Tree, Deciduous	3.85	1.04, 7.13	0.012	47.13	2.84, 1249.26
Tree, Conifer	16.83	5.67, 31.39	0.009	2.03e+07	289.27, 4.28e+13
Residential	2.20	-0.55, 5.34	0.137	9.02	0.58, 208.35
Grass, Non-maintained	3.58	0.63, 7.01	0.026	35.74	1.88, 1102.97
Spring/Summer Habitat III					
Intercept	-1.29	-2.28, -0.40	0.006	0.28	0.10, 0.67
Tree, Deciduous	2.33	0.32, 4.53	0.017	10.32	1.38, 92.81
Tree, Conifer	11.69	3.11, 22.72	0.028	1.20e+05	22.32, 7.33e+09
Grass, Non-	1.82	-0.10,	0.068	6.20	0.90, 47.72

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
maintained		3.87			
Spring/Summer Global Top Models					
Spring/Summer Global I					
Intercept	-2.22	-.74, -0.98	0.002	0.11	0.02, 0.38
Tree, Deciduous	1.58	-0.60, 3.85	0.065	4.84	0.55, 46.97
Tree, Conifer	10.32	1.15, 21.63	0.017	3.02e+04	3.16, 2.47e+09
Residential Low-density	2.62	-0.10, 5.60	0.018	13.76	0.90, 270.69
Park Low-use	2.73	0.91, 4.79	0.002	15.29	2.47, 119.94
FALL/WINTER HABITAT USE MODELS					
Fall/Winter Disturbance Top Models					
Autumn/Winter Disturbance I					
Intercept	-3.05	-5.48, -1.24	0.004	0.05	0.01, 0.29
Residential Low-density	5.88	2.43, 10.31	0.003	359.16	11.36, 2.99e+04
Park Low-use	4.36	1.76, 7.71	0.003	78.39	5.79, 2239.65

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
Autumn/Winter Disturbance II					
Intercept	-5.42	-9.94, -2.13	0.005	0.01	4.82e-05, 0.12
Residential Low-density	8.51	3.76, 15.05	0.002	4.94e+03	43.09, 3.43e+06
Residential High-density	4.09	-0.42, 9.28	0.090	59.84	0.66, 1.06e+04
Park Low-use	7.01	2.94, 12.41	0.003	1.10e+03	18.87, 2.46e+05
Autumn/Winter Habitat Top Model					
Autumn/Winter Habitat I					
Intercept	-24.90	-50.80, -8.24	0.021	1.53e-11	8.66e-23, 2.63e-04
Tree, Deciduous	26.09	8.84, 52.67	0.019	2.15e+11	6.91e+03, 7.50e+22
Tree, Conifer	28.28	8.92, 55.39	0.015	1.91e+12	7.47e+03, 1.14e+24
Residential	26.16	9.16, 52.32	0.017	2.30e+11	9.56e+03, 5.29e+22
Livestock	36.44	10.18,	0.043	6.70e+15	2.65e+04,

Parameter	Estimate	95% CI (+/-)	*P*-Value	Odds Ratio	95% CI (+/-)
		83.92			2.79e+36
Grass, Non-maintained	24.49	7.43, 50.74	0.026	4.36e+10	1.69e+03, 1.09e+22
Grass, Maintained	33.68	8.51, 70.74	0.031	4.21e+14	4.95e+03, 5.27e+30
Water	18.26	-3.07, 44.45	0.115	8.54e+07	0.005, 2.02e+19
Autumn/Winter Global Top Model					
Autumn/Winter Global I					
Intercept	-9.49	-17.52, -4.06	0.005	7.58e-05	2.47e-08, 0.002
Residential Low-density	13.30	6.07, 24.26	0.003	5.95e+05	435.36, 3.44e+10
Residential High-density	8.70	2.27, 17.30	0.020	5.98e+03	9.63, 3.25e+07
Park Low-use	11.09	4.93, 19.95	0.003	6.56e+04	138.49, 4.60e+08
Livestock	12.15	2.57, 27.18	0.037	1.89e+05	13.10, 6.37e+11

Chapter 5: Conclusion

Urbanization and human encroachment threaten wildlife through habitat fragmentation, increased disturbance, and vegetation alteration. As urbanization continues to spread (Brown et al. 2005), remaining 'natural-like' habitats within urban landscapes are particularly at risk from degradation associated with increased human presence and activities. Human-dominated land use alters existing habitat composition and predator communities which is a threat to many species (Marzluff and Ewing 2001, Shochat et al. 2006), perhaps none greater than ground nesting avian species. In addition, as human dominated areas continue to spread, human-wildlife interactions are becoming common place (Kellert 1985, Conover and Chasko 1985). While many interactions are positive in nature, wildlife living in close proximity can produce undesired results (Ditchkoff et al. 2006). To effectively manage these areas will require a greater understanding of diverse set of urban species' ecology.

This study examined the value of urban habitat for use by the wild turkey (*Meleagris gallopavo*). My research objectives centered on providing the first primary information on urban wild turkeys ecology, including: 1) assessing urban wild turkey nesting behavior and possible changes to reproductive measures, 2) investigation of urban wild turkey survival and the influence of local mortality agents, and 3) assessing urban wild turkey home range characteristics and habitat use. It is important to note that the results of this study should be used with caution as previous wild turkey research has

demonstrated that demographics and habitat use can vary greatly from one year to another and are highly dependent on local environmental conditions.

Wild turkeys have demonstrated an ability to use a diverse set of rural and agricultural habitats (Barwick and Speake 1973, Little and Varland 1981, Vander Haegen et al. 1988, Thomas and Litvaitis 1993, Wright et al. 1996, Miller et al. 1997, Thogmartin and Schaeffer 2000, Wright and Vangilder 2001, Hubert 2004, Wilson et al. 2005, Humberg et al. 2009); however, our knowledge regarding the species response to increased levels of disturbance, habitat alteration, or resource availability associated with urbanization remains limited. By examining the wild turkey's use of urban landscapes, I provide a basic understanding of the species urban ecology, from which we may draw conclusions not only on management strategies for urban wild turkey populations, but also for ground nesting avian species in general.

I found observed wild turkey nesting traits and nest survival across all study areas were similar to published literature for rural or agricultural wild turkey populations. Further, most nesting measures did not differ by study area or age-class, with average clutch size and date of incubation being the two exceptions. I found clutch size differed with urban intensity (Tinsley 2014*a*). This is an important consideration as it may indicate a response to possible changes in urban predator communities or changes to vegetative structure. I found date of incubation differed with age-class, which may suggest a lack of suitable habitat or experience as juveniles, on average, began nesting activities later than adults.

Based on my data and observations, vegetative characteristics related to nest site concealment and proximity to open water were primary factors influencing wild turkey nesting ecology in the urban setting (Tinsley 2014*a*). For this study, females tended to locate their nests in habitat with greater concealment. This is a common characteristic observed for rural wild turkey populations (Lazarus and Porter 1985, Holbrook et al. 1987, Badyaev 1995) and other ground nesting bird species (Martin 1993). Presumably vegetative features assisting in nest site concealment also act as a barrier predators and shelter from climatic conditions.

The importance of water resources in proximity to nesting wild turkeys is unclear. I found wild turkey nesting attempts had a greater chance of success if the nest was located at a greater distance from open water (Tinsley 2014*a*). I hypothesis for wild turkey populations occurring on urbanized areas, open water avoidance is likely linked to possible changes in local predator communities. For example, if local conditions alters the density or composition of ground nesting predators associated with water, then nesting success should favor females located a greater distance from open water. This result, combined with the lack of significant influence of local vegetation on species nesting success, suggests further study of suitable habitat in urban habitats is essential to determine whether changes to land management practices could enhance habitat suitability for urban ground nesting birds.

As in other areas of the wild turkey range, weather likely had a strong influence on observed wild turkey reproduction. Previous research has clearly demonstrated adverse weather conditions impact female condition, the reproductive cycle (Porter et al. 1983), and poult survival. Research completed in northern climates demonstrates rural wild turkey populations often rely on agricultural operations for supplemental dietary resources. My rural birds were no exception, with most monitored birds and other wild turkey observations consistently associated with agricultural areas during winter. However, urban and suburban wild turkeys in this study did not have access to large-scale agricultural operations. Instead, I observed the birds using available resources, such as bird feeders, food plots, acorns, or fruiting trees (i.e., various crab apple specimens, domestic apple).

I found predation was a leading source of mortality among female wild turkeys regardless of urban intensity (Tinsley 2014*b*). My results are consistent with rural wild turkey literature, which tends to list predation as a leading cause of fatality (Vangilder and Kurzejeski 1995, Miller et al. 1998). In rural literature, predator control is often suggested as a way to alleviate pressure on wild turkey populations; however, as with other species these control measures are often short-term and not cost-effective (Beasom 1974, Duebbert and Kantrud 1974, Speake 1980). Instead as the majority of fatality occurs during the incubation period (Tinsley 2014*b*), I recommend concentrating on habitat features beneficial to nest concealment.

Our understanding of suitable wild turkey habitat has continued to evolve since conservation and species management began. Early wildlife managers believed wild turkeys required large tracts of forested landscapes and absence of human presence. These early habitat assumptions were based largely on the fact that remaining wild turkey populations were only found in remote forested areas. Since then, wild turkeys have proven to be much more adaptable, using a diverse set of rural habitats.

My research supports the viewpoint that wild turkeys, specifically eastern wild turkey populations in northern climates, require much less forest habitat than once believed. As expected, I found most female wild turkeys used available 'natural-like' habitats for reproductive activities, seeking remote and low disturbance areas (e.g., grass along highway medians, remote areas of urban park systems) in urbanized landscapes (Tinsley 2014*c*). Interestingly, despite the importance of more natural-like habitat for female urban wild turkeys, most individuals exhibited a high reliance on developed, human-dominated areas during autumn and winter, despite the species reputation for intolerance of human activity (Tinsley 2014*c*).

Although urbanization creates a diverse set of habitat patches, the success of existing urban wild turkey populations provides some level of optimism for the potential of urban habitats to provide suitable habitat for ground nesting birds. Future research must include long-term monitoring of a diverse set of urban habitats and ground nesting species. This, ultimately, will provide greater insight into avian responses to a broader range of urban

vegetative and predator conditions. Moreover, we need demographic data to establish whether urban ground nesting birds are successfully reproducing in the urban environment, and potential changes that may occur. Furthermore, we require a greater understanding of the impact human disturbance has on urban wildlife populations (Chace and Walsh 2004). Pressure to manage our remaining natural areas for multi-purpose recreational activities will increase human disturbance along the perimeter and within these habitats. Long-term monitoring will support increased species sampling, provide sufficient yearly data to evaluate the influence of temporal variation, and enable the assessment of the value for each urban habitat type to provide suitable habitat for urban ground nesting birds.

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